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Key physical and chemical drivers of fisheries productivity (flow, nutrient and thermal regimes) across rivers in various Canadian regions: lessons learned from NSERC's Hydrofiet 2010-2015

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

This report summarizes findings of general interest under the physical habitat research theme of NSERC-HydroNet (2010-2015). The report addresses 3 key, physico-chemical determinants of fish habitat productivity below hydro dams: the degree of alteration of the natural flow regime (Section A of this report); the river's nutrient regime (Section B) and the alteration of its thermal regime (Section C). Each of the three self-contained sections briefly reviews key literature and then identifies the specific objectives of HydroNet work on the particular set of determinants of habitat productivity that it addresses. Then each section describes some low cost multi-site field sampling and statistical modelling approaches that were used to efficiently monitor the effects of regulation, in turn, on a particular river system's flow regime, nutrient regime and thermal regime. Following this, each section presents significant patterns that emerged from comparatively analyzing aspects of the large data sets collected under HydroNet, covering a wide range of regulated and reference river systems across Canada. In particular, some specific effects of Run of the River, Peaking and Storage type hydro systems on both flow and thermal regimes are illustrated. Finally, the analysis demonstrates the important roles played by variations in riverine nutrient levels and species richness in determining total fish biomass, a useful index of habitat productivity in each system.

Principaux facteurs physiques et chimiques de la productivité des pêches (débit, nutriments et régimes thermiques) dans des cours d'eau de diverses régions canadiennes : leçons tirées du réseau HydroNet du CRSNG 2010-2015

RÉSUMÉ

Le présent rapport résume les constatations d'intérêt général associées au thème de recherche de l'habitat physique du réseau HydroNet du Conseil de recherches en sciences naturelles et en génie du Canada (CRSNG) 2010-2015. Le rapport porte sur trois principaux facteurs physico-chimiques déterminants de la productivité de l'habitat du poisson en aval des barrages hydroélectriques : le degré de modification du régime d'écoulement naturel (section A du présent rapport); le régime nutritif du cours d'eau (section B), et la modification de son régime thermique (section C). Chacune des trois sections autonomes examine brièvement les principales publications scientifiques, puis détermine les objectifs particuliers des travaux sur le réseau HydroNet en ce qui concerne l'ensemble précis de facteurs déterminants visés de productivité de l'habitat. Ensuite, chaque section décrit certains des échantillonnages à faible coût sur le terrain à plusieurs endroits et des approches de modélisation statistique qui ont servi à surveiller efficacement les effets de la régulation sur le régime d'écoulement, le régime de nutriments et le régime thermique d'un réseau de cours d'eau en particulier. Par la suite, chaque section présente les tendances importantes qui ont ressorti de l'analyse comparative des aspects des grands ensembles de données recueillis par le réseau HydroNet, couvrant un large éventail de systèmes de cours d'eau réglementés et de référence au Canada. Plus particulièrement, certains effets précis des systèmes de dérivation de cours d'eau, de demande hydroélectrique de pointe et de stockage de l'eau sur les régimes d'écoulement et thermique sont illustrés. Enfin, l'analyse démontre le rôle important joué par les variations de l'apport en nutriments et la richesse des espèces fluviales pour déterminer la biomasse totale du poisson, qui constitue un indice utile de la productivité de l'habitat dans chaque système.

SECTION A - FLOW REGIMES: A tool to assess the types and relative severity of flow regime anomalies associated with particular hydro dams

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OUTLINE: WHAT IS THE TOOL AND WHAT IS IT USEFUL FOR?

When regulators analyze new hydro projects (or recertify existing installations), they need to assess the possible severity of the project's flow regime modification, among other 'pathways of effects' on fish populations. To this end, an "ideal" (but data expensive) approach would involve the use of empirical relations specifying Ecological Limits of Hydrological Alteration (ELOHA), calibrated to the particular region and its aquatic faune (Poff et al. 2010). Empirical ELOHA relations are based on detailed, multi-site data on both altered flow regimes and biota. They aim to quantify, for various geographic classes of river ecosystems, the thresholds of flow alteration leading to unacceptably altered levels of biotic metrics of interest, including measures of fish population health (Acreman and Dunbar 2004; Arthington et al. 2006). Unfortunately, the data gathering in Canada remains to this day insufficient to populate and evaluate such ELOHA models. However, the HydroNet 2010-15 data base on fish communities and flow regimes provides a pilot feasibility demonstration in this direction (Boisclair et al. 2016; Smokorowski et al. 2011).

In the current absence of fully tested ELOHA rules, less powerful approaches are available to assess flow regime alteration. Building on previous work of McManamay et al. (2012 a, b,c), HydroNet 2010-2015 developed one such approach for Canadian conditions. Illustrations of its use (along with key methodological details) are given in this section of the report. Further details are given in McLaughlin et al. (2014). In contrast to ELOHA, the approach presented here does not identify ecological thresholds of allowable alteration; however it can yield a ranking of regulated systems in terms of both the type and the relative severity of their flow regime **anomalies** compared to unregulated norms. These data can be used to rank the severity of flow regime alterations across a set of projects (past and proposed) and to prioritize particular regulated systems for more detailed impact studies. The office based method presented here can be applied, simply using readily available hydrometric data sets, to any regulated site of interest, in any region, and at any level of detail justified by the project.

The main aim of this section (A) of the report is to demonstrate some potential uses of this practical approach to quantifying regulation effects on natural flow regimes.

DEFINITIONS: FLOW REGIME ALTERATIONS AND FLOW REGIME ANOMALIES: DIFFERENT CONCEPTS FOR DIFFERENT USES

1. Here we define a **flow regime "alteration"** as the change of various aspects of the flow regime caused by hydro operations. The degree **of flow regime alteration** can thus be quantified by directly comparing on the given system various "flow metrics" (such as mean annual flood, mean January flow, etc.) computed over pre- and post- damming flow periods (Richter et al. 1996, 1997). This requires that multi-decade long flow records exist for both pre- and post- regulation. It also assumes that climate and watershed hydrology both remained stationary (unchanged) over both pre and post regulation periods (or alternatively that we can identify and remove longer term climatic and land use trends modifying flow regimes). Unfortunately, for many existing hydro-sites in Canada, pre-regulation flow

records are often too short to carry out the required Before-After comparison (this indeed was the case at most selected Hydrofiet regulated study sites).

More importantly, even when sufficient historical flow records exist, this technique does not, on its own, clarify to what extent the altered (After) flow regime is anomalous compared to the range of natural regimes to which are exposed fish populations in other watersheds in the region, which is the main application of the alternate approach proposed here.

2. The calculation of scores of **flow regime anomaly**, presented in this report, avoids these data gap problems and provides an alternate and conceptually distinct approach to assess the degree of severity of flow regime alteration at a regulated site. Anomalies are quantified ("scores" are generated) by comparing the post-damming regulated regime to a range of flow regime "normals" observed at unregulated sites with broadly comparable watershed character, located in the same hydrological region (McManamay et al. 2012 a,b,c; McLaughlin et al. 2014). This approach does not require pre-regulation flow data. Instead it only requires analyses of 2-3 decades of recent hydrometric records, at a set of regulated and otherwise broadly comparable unregulated sites in the study region. This analysis can be conducted with existing data (or in the case of proposed developments using simulated future hydrographs), at practically all existing or proposed hydro sites in any region of Canada.

3. We illustrate below how the **anomaly scores** generated here are a practical way of detecting and quantifying various types of regime anomalies. We first illustrate how simple visual inspection of hydrographs is a poor tool to detect subtle but potentially important features of the regulated flow regimes. In contrast, the statistical method presented here yield scores for various **anomalous** features of the studied flow regime (i.e. individual anomaly scores for various metrics of flow magnitudes, durations, timing, etc.). Combined with expert knowledge on fish ecology, the size of anomaly scores can then provide a simple 'heuristic' to estimate the probability of significant ecosystem impacts at regulated sites.

   Specifically, the tool can identify those regulated systems with the strongest anomaly scores and thus help prioritize these sites for more detailed monitoring of possible faunal impacts. The analysis can also disclose where potentially altered flow regime conditions at some regulated sites remain well within the normal range of variability for unregulated (reference) sites in the region. In such cases, if healthy fish populations are documented at unregulated sites with a comparable set of flow metrics as those computed at a given regulated site, this finding can support claims that the flow regime alteration caused by regulation may be, in itself, too minor to produce negative impacts on fish fauna.

4. It is important to note that regime **anomaly** and regime **alteration** as used in this report are different concepts that serve different purposes. A regime "anomaly" as defined here, in theory, does not always represent a flow regime alteration due to regulation. Some other highly anomalous watershed characteristic of the regulated system, atypical for its geographical location and flow class (such as unusually large lakes or expansive urban areas) could also be at play, irrespective of regulation. Conversely, regime alteration by hydro damming does not always produce a clearly anomalous regime (for example, weekly regulated systems with little hydro-reservoir storage and no peaking activity may behave quite similarly to unregulated systems draining small, upstream lakes). However, all regime anomalies detected by the proposed method across the set of 16 Hydrofiet regulated sites analyzed here were related to hydro-regulation effects.

INTRODUCTION

Each riverine ecosystem is unique and dynamic. First, a river's flow regime is known to affect the ecosystem's physical structure (channel widths, depths, bed sediment sizes and seasonal sediment stability, water turbidity and temperatures, etc. (Church 2002; Cassie 2006). By controlling bioenergetics as well as spawning, feeding and refugia habitat conditions, the flow regime is also an important driver of the river system's ecological integrity (Booker et al. 2007; Bunn and Arthington 2002; Lytle and Poff 2004; Pluckridge et al. 1998; Poff et al. 1997; Poff and Zimmerman 2010; Mims and Olden 2012). The ecologically important features of a river's flow regime are summarized by sets of hydrologic indices (or "metrics") that describe the magnitude, frequency, duration, timing and rate-of-change of flow events (Poff et al. 1997). The most widely used sets of indices are the Nature Conservancy's Indicators of Hydrologic Alteration (IHA) (Richter et al. 1996) and the United States Geological Survey's (USGS) Hydrologic Index Toolkit (HIT) (Olden and Poff 2003), which includes the IHA.

After construction of a storage reservoir, downstream river flows can be artificially modified by dam operators, within project design limits. The degree of flow alteration that occurs is quite variable however, as it depends on the physical characteristics of the impoundment (reservoir volume and outlet structures can limit the amount of water that can be retained or quickly released) and its mode of operation (e.g. number of turbine units used in peaking and their flow capacities, environmental release rules set by regulators, etc.). To generalize, research shows that hydro dams with significant storage volume (or water residence time) tend to be used in ways that reduce maximum flows, reduce overall flow variability and rise rates (excepting peaking systems which increase hourly change rates), while very often increasing minimum flows and the number of flow reversals per year (Magilligan and Nislow 2005; McManamay et al. 2012a; Poff et al. 2007). However, specific alterations to the natural regime are quite variable across dammed river systems. Various approaches can be used to quantify the type and intensity of regime modification at a particular site.

Regime alterations (*cf.* definition 1 above) are conventionally identified by comparing flow indices calculated from pre-dam flow records to indices calculated from the post-dam, regulated records (Richter et al. 1996). However, where pre-regulation flow records are too short (under 2 decades) for minimal statistical validity, this analysis can become complex and imprecise. Given insufficient pre-regulation records, alteration estimates may require hind-casting of pre-dam discharges using complex runoff hydrology and flow routing models, possibly requiring pre-damming weather and land use data to account for climate and land use changes pre vs post regulation.

However, scientists and managers can be interested in comparing the degree and type of flow regime characteristics across numerous dammed systems, even in the absence of historical flow data. Thus a need for an alternate methodology focusing on the detection of **flow regime anomalies** (*cf.* definition 2, above) has been proposed (McManamay et al. 2012 c). This approach first requires identifying classes of unregulated rivers that share broad hydrologic properties with the pre-regulated state of the dammed system. McManamay et al. (2012a) describes flow classes as "groups of streams that share similar natural hydrology; thus, they provide a stratified approach to evaluate hydrologic alterations or departures from natural baseline conditions." Analyzing the range of flow regime traits within each reference flow class, the natural range of variability of various hydrograph components can be determined, and significant deviations in regulated systems can be identified by comparison (Poff 1996; Poff et al. 2006; McManamay et al. 2012a). Such an approach is illustrated here with application to Canada.

Our specific objectives under the Flow Regimes component of the Hydrofax (2015-2016 NSERC-SHG Program were:

1. to classify 95 unregulated rivers proximal to 19 regulated Hydrofax rivers for which high-quality daily flow records were available into 'flow classes' representative of regional flow regime norms, and to evaluate the main qualitative differences between these flow classes;
2. to assign each of the 19 Hydrofax regulated rivers spread across Canada to the appropriate flow class based on geographic location and physical watershed characteristics (pre-impoundment);
3. to compute metrics of flow regime anomalies for each regulated system evaluated by comparison with the range of metric values normally observed in comparable unregulated systems in the appropriate flow class;
4. to describe the main types of observed regime anomalies across our Canadian study set as a function of regulation type (large storage dams versus run-of-the-river vs peaking systems).

A variety of results that illustrate the method are given here (more detail is given in Wu-Laughlin et al. 2016, and in Wu-Laughlin 2016).

STUDY SITES

For logistical and budget reasons, Hydrofax targeted its data gathering on a subset of small-to-medium sized southern Canadian rivers deemed for hydropower, with watershed areas mostly in the range 500-2000 km^2 (median value, approx. 1000 km^2). Site selection was strongly conditioned by ease of river site access and feasibility of accurate fish sampling by electrofishing and seining. (The fish sampling methods and resultant data are summarized in other Hydrofax 2015-2016 Hydrofax reports, specifically see Brezner et al. 2016 and publications referenced therein). Hydrofax site selection was also stratified by region (with a cluster of sites in the Western Cordillera of Eastern BC and western Alberta, a second in Ontario and Quebec Shield regions, and a third in southern Quebec and New Brunswick Appalachian geomorphic regions). Finally Hydrofax sought to include sites with various types of regulation (large storage dams with or without peaking operations as well as Run of River or RORs). Based on quality of flow records, 19 regulated rivers across these regions were finally selected to study impacts of hydro operations on flow regimes. All were dammed for hydro-power production between four decades and a century ago.

In the next step, we identified as potential reference sites in each region, the largest possible number of nearby river gauges located on unregulated watersheds within a range of half to twice the watershed size of each regulated site. For this comparative analysis, the distance required in general to secure a sufficient number of reference sites of similar drainage area will depend on the watershed size of the regulated sites being studied. In this study, with regulated sites of drainage area 500-2000 km^2, all possible reference gauges within a radius of about 400-500 km from the regulated site were identified. This yielded in the range of 5-20 reference sites per regulated site from which we could establish regional flow class 'envelopes'. 95 reference gauges were identified this way, to contextualize natural flow regimes relevant to comparisons with the 19 regulated sites located in the various sub-regions studied.

Other than discarding gauges identified as draining regulated watersheds or watersheds incorporating large cities, no attempt was made to target or reference sites only to certain 'pristine' watersheds (nonexistent in our many regions). Nor could we target for reference sites specific narrow ranges of watershed detail, % lake cover, or % forested lands, closely matching conditions at specific regulated sites. Thus the range of flow regimes observed in each

Figure 4A. [illegible]

[illegible]

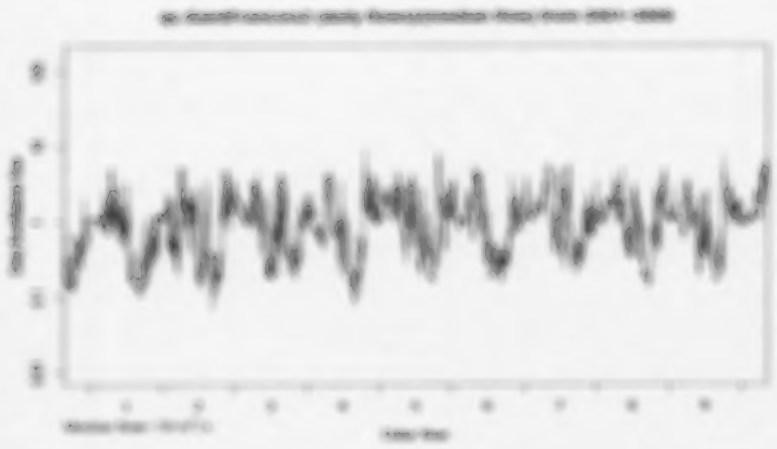

Figure A3: Nine recent years of hydrographs at Saint-François River site 2, a regulated site.

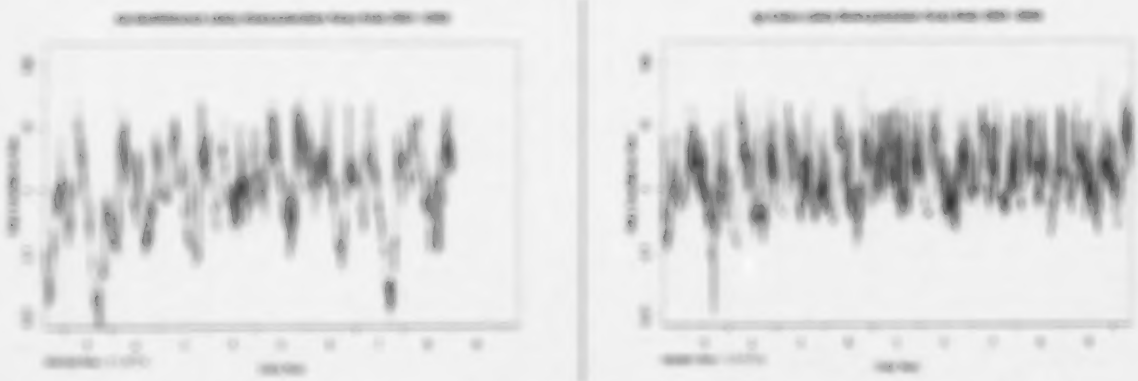

Figure A4: To illustrate the difficulty of visual comparisons, the same nine years of hydrographs at 2 reference sites from class E2, the region class to which the unregulated Saint-François River was assigned. All discharge values are normalized by the median daily discharge at each gauge. The quantitative anomaly comparisons conducted next compare metrics at Saint-François site2 to those extracted over 18 such reference hydrographs, all belong to class E2 (cf Figure A5).

Figure A3 illustrates the basic idea behind anomaly detection. It plots the values of only 2 metrics (shown on the axes) for three regulated systems (red dots) belonging to E2, including the Saint-François 2 site shown in Figure A3, against the same metrics for the 18 unregulated, reference systems in E2 (blue dots). The plots reveal that the Saint-François 1 and 2 sites have a much lower values of flow volumes during high runoff season (on X axis, 3 month scale mean discharge values, here again scaled as multiples of median daily flow) than are observed among the entire E2 flow class in southern Quebec and northern NB. They also have much lower values of the coefficient of variation of daily flows (shown on Y axis).

Figure A3 further shows that both Saint-François regulated sites have broadly similar degrees of anomaly on these metrics, measured by a distance to the centroid of the metric values across the 18 reference rivers (which are the blue dots). The 2 same metrics for the regulated Etchemin River (red dot in center of group) are seen to be non-anomalous, compared to the

mean and variability within the reference group. The Etchemin was classified for the purpose of HydroNet as a Run of the River (ROR) system: its storage volume is quite small relative to mean annual flow and all its metrics show values that are within the normal range of variability for the set of E2 reference flow regimes.

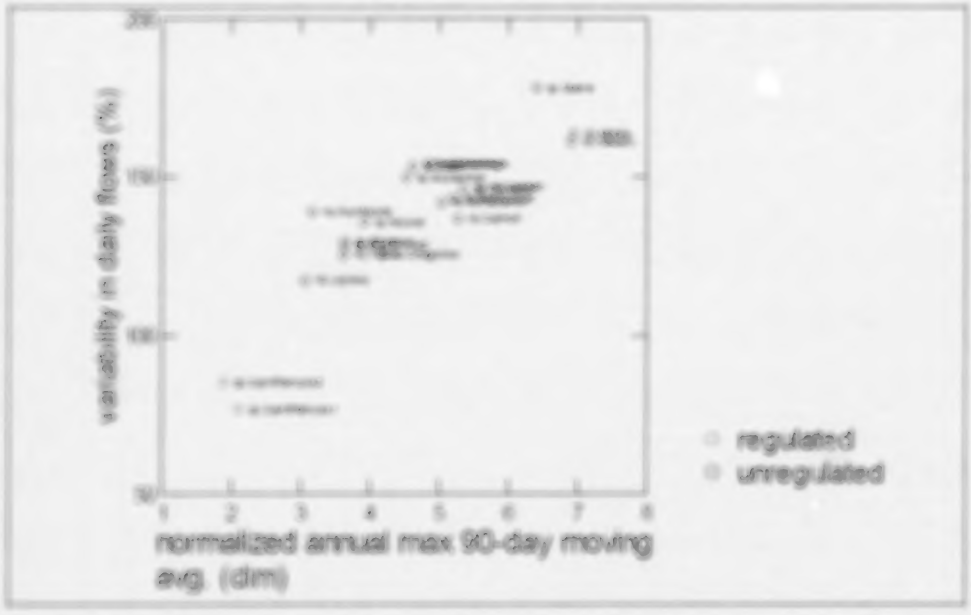

Figure A5. This plot displays 2 (nDHS vs. MA3) of the top 4 loading indices on PC2 across flow class E2, the gradient of range of variability on which the two Saint-François sites display most anomalous behavior. It is apparent from this plot (and also true for the other high loading PC2 indices that we explored) that the Saint-François has systematically reduced index scores for the indices that highly influence PC2.

We next generalize to all 70 metrics retained in this study the comparisons shown in Figure A5. This is done by calculating Z-scores of degree of anomaly (Z-scores) for all 70 metrics for the Saint-François regulated site (recall from Methods that the Z-score for a regulated site metric represent the number of reference class (E2 here) standard deviations that its value lies above or below the reference class mean for that same metric). For the regulated and reference rivers in class E2, Principal Component Analysis of all 70 Z-scores can then be used to visualize (Figure A6) the distribution of regulated versus reference score values, by projecting them on the main axes of variability across the reference rivers of the set of 70 metrics.

This plot confirms that while the Etchemin river has similar scores to unregulated rivers on the two main PC axes of metric variability among reference rivers, the Saint-François sites plot unusually low (i.e. display regime anomalies) on PC2 (Y axis) a combination of various metrics describing high flow, spring time runoff volumes, one of the main axes of variability across E2 unregulated sites. Figure A6 also shows however that the Saint-François sites do not display anomalous behavior in terms of metrics of frequency of low flows nor flashiness (on PC1, X axis, the other main axis of natural variability among E2 class rivers).

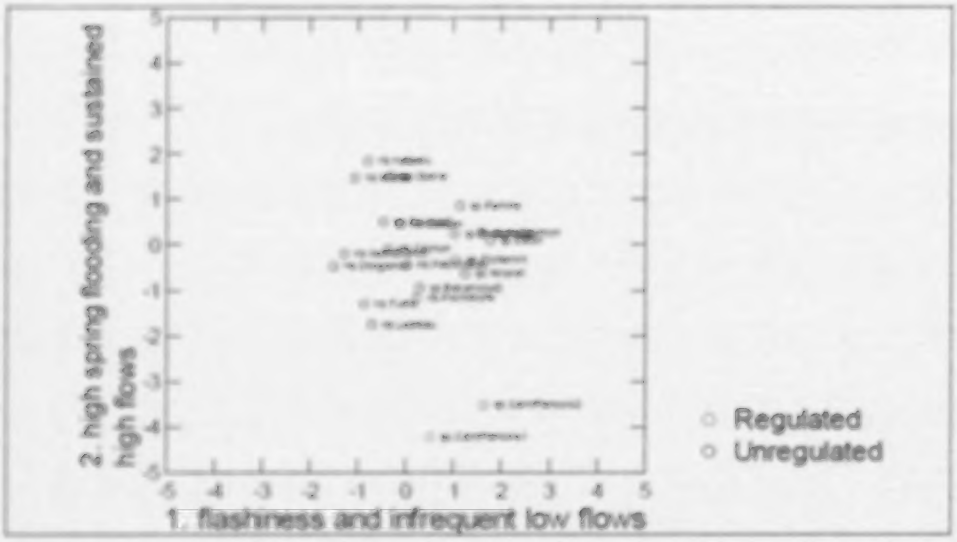

Figure A6. Regulated and unregulated flow class E2 rivers in multivariate PCA space based on all unregulated rivers of this class. PC1 explains 29.5% of the total variance within E2 reference sites. PC2 explains 19.2% of the total variance. Here PC1 and PC2 scores have been standardized based on the reference set means and SD. If regulated rivers plot well outside the cluster of unregulated reference systems along a PC axis, certain flow indices loading this axis are highly anomalous. As conventional, axes are named based on an evaluation of the top loading indices. For details see McLaughlin et al 2014. Note that the Saint-François River plots outside the range of reference scores on PC2 while the Etchemin River falls within the reference range on both PC1 and PC2.

Pinpointing the most anomalous particular features of the flow regime at a regulated site.

For any given regulated system, anomaly scores can be used to identify, and target for further enquiry, particular features of a regulated flow regime that are the most anomalous (highest Z-scores) and thus possibly most likely to be limiting for particular species within the regional fish fauna (or, in some cases, most likely to favor invasive fish species). This information can then be interpreted through the lens of the expert judgment of fish population biologists with knowledge of regional fauna.

Table A1 presents a subset of the 70 flow regime metrics at the Saint-François 2 site that are most highly anomalous (absolute Z-score greater than 3.5, that is the regulated metric is more than 3.5 times the standard deviation away from the reference mean for that same metric). Similar tables have been generated and discussed for the 16 regulated HydroNet study rivers (McLaughlin, 2014). For each metric listed here, the third column gives its anomaly score (Z-score). In the cases where the reference metric distributions are normal, the second column also gives the probability that such a Z-score at the regulated site is consistent with the null hypothesis (i.e. the metric value remains consistent with the range of normal variability observed within its reference class).

Table A1. The most anomalous flow metric values observed at the regulated Saint-François River site 2. Column 3 gives the metric's Z-score (i.e. the number of standard deviations above or below the class mean for the given metric within reference class E2). The name and a definition of each metric are given in columns 6 and 7. Means and standard deviation values for each metric across the reference set of unregulated rivers in flow class E2 are given in last 2 columns.

Index Code	p-val	z-score	load1	load2	Index Name	Description	Units	Raw Value	Class Mean	Class SD
RAS	0.000	**8.54**	0.54	0.40	Proportion of rise days (Number of day rises)	Compute the number of days in which the flow is greater than the previous day. RA5 is the number of positive gain days divided by the total number of days in the flow record.	dim	0.42	0.28	0.02
MA41	0.000	**7.41**	0.41	-0.23	Specific mean annual runoff	Compute the annual mean daily flows. MA41 is the mean of the annual means divided by the drainage area.	m^3s^{-1}/km^2	0.10	0.03	0.01
nML1	0.000	**5.98**	-0.45	-0.51	Mean minimum monthly flows (Jan)	For each year, compute the minimum flow for the month on January. ML1 is the mean (or median – Use Preference option) of these values.	dim	1.01	0.47	0.09
DA4	0.000	**-4.48**	-0.65	-0.32	Median fall period	Median duration of consecutive periods of falling values.	days	2.00	4.11	0.47
MA6	0.002	**4.13**	0.41	-0.50	Ratio of low flow (Q90) to high (Q10) quantiles	Range in daily flows is the ratio of the 90 percent to 10 percent exceedance values for the entire flow record (low flow/ high flow). Compute the 90 percent and 10 percent exceedance values for the entire flow record. Exceedance is computed by interpolating between the ordered (descending) flow values. Divide the 90 percent exceedance value by the 10 percent value.	dim	0.12	0.04	0.02
FH3	0.003	**-4.04**	-0.01	0.81	Days per year above 3x flow median	Compute the number of days each year that the flow is above a threshold equal to three times the median flow for the entire record. FH3 is the mean (or median – Use Preference option) of the annual number of days for all years.	days/year	17.00	73.36	13.97
nML3	0.004	**3.99**	0.31	-0.68	Mean minimum monthly flows (Mar)	For each year, compute the minimum flow for the month of March. ML3 is the mean (or median – Use Preference option) of these values.	dim	1.01	0.56	0.11
MA3	0.006	**3.88**	0.20	0.90	Annual Variability in daily flows 1	Mean (or median – Use Preference option) of the coefficients of variation (standard deviation/ mean) for each year. Compute the coefficient of variation for each year of daily flows and then compute the mean of the annual coefficients of variation.	%	84.99	144.24	15.28
ML17	0.006	**3.87**	0.37	-0.46	7-day minimum baseflow	Compute the mean annual flows. Compute the minimum of a 7-day moving average flow for each year and divide them by the mean annual flow for that year. ML17 is the mean (or median – Use Preference option) of those ratios.	dim	0.16	0.06	0.03
RAS	0.021	**3.57**	0.59	-0.32	Number of reversals (day-to-day)	Compute the number of days in each year when the change in flow from one day to the next changes direction. RAS is the average (or median – Use Preference option) of the yearly values.	events/year	128.50	88.94	11.09

Generating hypotheses concerning "pathways of effects" on fish, based on anomalous Z-scores.

Useful insights on potential fish population impacts can be generated from these statistical analyses. Resulting impact hypotheses need then to be further investigated by project biologists. Some possible effects are described next for the case of the regulated Saint-François River.

It was observed above (Figure A5) that 3-month spring high flow volumes on this system are unusually low compared to its reference group E2. This might have effects on the duration of pike spawning in the riparian vegetation along the Saint-François river floodplain. Table A1 also shows that the strongest absolute anomalies on the Saint-François compared to its regional references (Z-scores over 4 in Table A1) relate to the constancy and high level of both early and late winter discharges. The January minimum daily flow (in row 3, scaled to median flow always) is almost 6 standard deviations greater than the mean for the reference set E2. This is a typical effect for a base load generating system with large storage dam in Quebec. Such a regime anomaly may have effects on mid-winter ice cover formation in steeper river reaches and thus possible effects for winter habitat conditions of some fish. Note also that the March minimal flow levels are also 4 standard deviations higher than normal for the reference E2 group, possibly accelerating breakup in lower gradient river reaches where an ice cover has been able to form. Also notable (last 2 rows of Table A1) are the 7 day minimum flow in summer which has a Z-score of 3.9 ($p= 0.006$), likely implying (compared to reference group conditions) more moderate water temperatures during heat waves and less thermal stress for cool water species. Finally, the number of days with flow reversals (likely caused by release operations) at 128 (Table A1), is 3.6 SD higher than expected among the 18 reference rivers (which has mean of 89, SD of 11 days, Table A1).

Some general trends on the effects of mode of regulation, observed from pooling all HydroNet study sites across Canada

The analyses presented above allow the comparison, within one region and flow class, of the mode and intensity of regime "anomalies" across various regulated sites. This general approach can also be used to compare regime anomalies across a larger set of regulated rivers, pooling sites across different regions and different reference classes.

Figure A7 presents an ordination of anomaly (Z) scores for all 70 studied metrics across all 16 HydroNet regulated sites, distributed from BC to New Brunswick. (Each regulated site's Z-scores are computed with respect to its own appropriate reference flow class, as the basis for anomaly determination). This analysis reveals the qualitatively different effects of various types of regulation across the HydroNet sample. As expected, peaking systems (in red) tend to strongly increase the strength of day-to-day flow variations (flashiness) compared to relevant regional norms. In contrast, storage systems (in green) mostly tend to increase baseflow related metrics. Finally, ROR systems (in blue) have low Z-scores (behaviors similar to applicable reference class norms). In this ordination space, ROR scores lie within the cloud of scores (blue ellipse) for the unregulated references. The 2 New Brunswick sites (in purple) have unusual regimes compared to the rest of the study set. They appear to be operated to store headwater runoff for occasional but irregular rapid ramping of releases, lasting many days or weeks, to supply hydro dams further downstream. The differences in resulting flow regime when compared across all 70 selected metrics to more typical daily peaking regimes (red dots) stand out clearly in the anomalies analysis.

Figure A7 also illustrates how this approach can be used to rank regulated systems across Canada in terms of the overall type and degree of regime anomaly, expressed as the 2D

distance and direction of a site scores on such PC plots distance from those of unregulated references (blue ellipse here). Using this metric, it can be noted that among the small sample (5) of peaking sites (in red) studied under HydroNet, one site, the Kananaskis River site 2 in Alberta (below Pocaterra Dam), displays the most anomalous regime. Compared to Kananaskis 2, the peaking regime observed on the Magpie and Michipicoten Rivers in Ontario, when compared to the norms of its local reference class, is relatively more benign in terms of flashiness (PC1) although it has a greater homogenization effect (PC2) on seasonal flows (smaller range of magnitude and higher baseflows).

Across HydroNet study storage sites (in green), the Saint-François 1 site (bottom right on Figure A7) is seen to display the strongest degree of seasonal homogenization of regime (the lowest range of magnitude on PC2 axis, compared to its regional references) while the much smaller reservoir at Sugar Lake on the Shuswap River in BC produces much a weaker homogenization anomaly. (This type of ranking information, across any regime axis, may be useful for site selection towards further studies aiming to establish thresholds of flow anomalies compatible with acceptable impacts on fish populations. Note that such "euclidian distances" from class means can alternatively be computed using z-scores across all 70 anomaly metrics, instead of just using summary PC scores as illustrated on Figure A7).

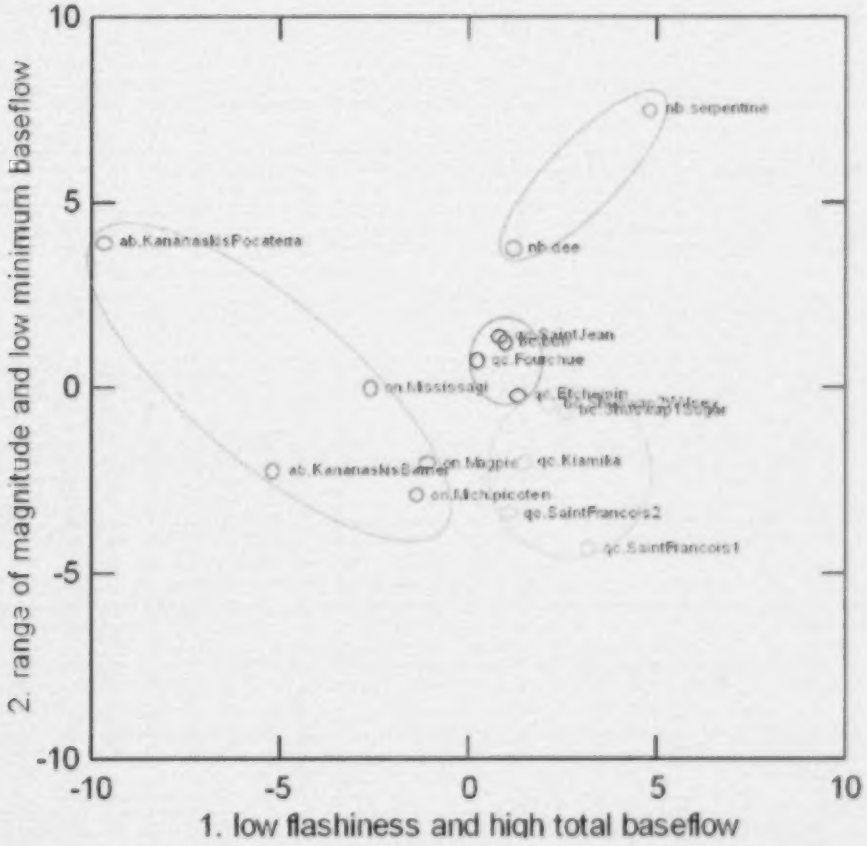

Figure A7. Principal Component analysis of Z-scores for all 70 studies metrics across all 16 regulated study sites in Canada (here each Z score is computed against the mean and SD of the given metric for the appropriate reference flow class at each site). Color codes refer to types of regulation: red for peaking sites, purple for atypical sudden release modes, green for mainly storage effects and blue for Run of the River sites (see discussion in text).

Finally, pooling across all 16 HydroNet regulated sites, Figure A8 presents histograms of anomaly scores (Z-scores) for all 70 flow regime metrics, grouped by regulation type, each overlain on Z-score distributions for all relevant reference (unregulated) sites. In aggregate, Figure A8 shows that peaking systems in the HydroNet sample display the largest proportion of the highest anomaly metrics (with Z-scores indicating values of particular metrics in some cases more than 10 or even 20 reference group standard deviations away from class means), followed by storage systems. See McLaughlin 2014 for more detail on the nature of these various types of regime anomalies and a discussion of possible effects on fish populations of various types of regime anomalies, based on literature reviews.

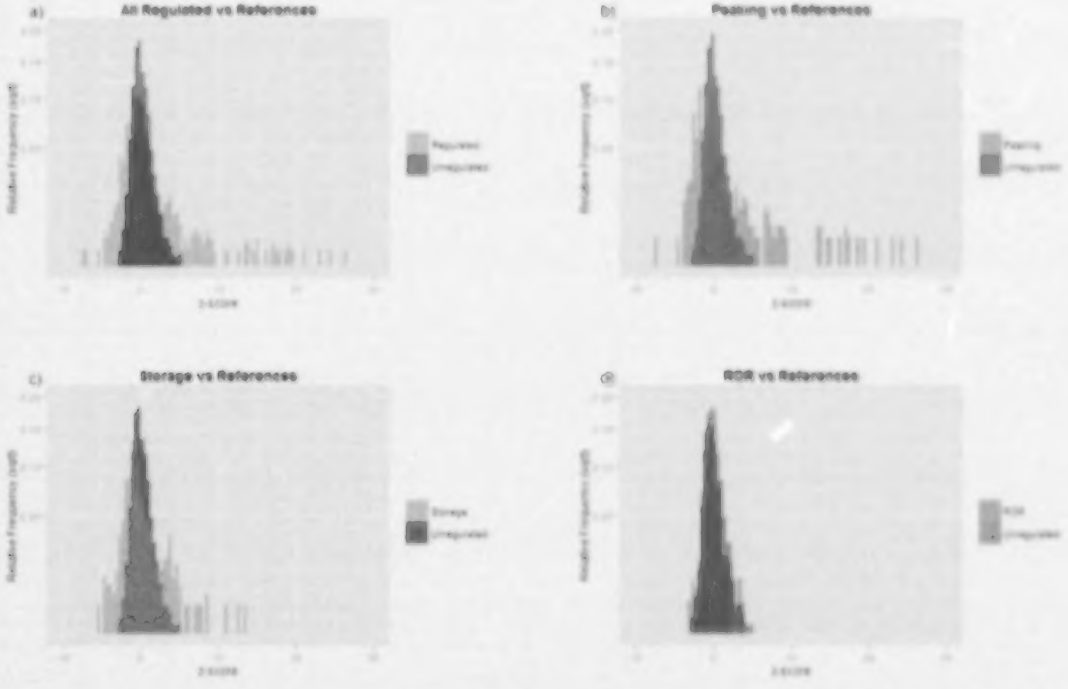

Figure A8. Histograms of all 70 anomaly scores observed over the 16 regulates sites studied across Canada, compared to histograms of reference site scores (i.e. scores at unregulated sites). Colored plots are histograms for subsets of the 16 sites classified by regulation type, overlain on these sites reference scores: b) peaking sites, c) storage sites and d) ROR sites.

CONCLUSIONS

It is important to recall that the purpose of the approach presented here is not to quantify the precise degree of **alteration** (see definition 1 given in this section's Outline) of the flow regime following damming (Richter et al. 1996 describe approaches to this end). Rather the aim is to clarify the degree of **anomaly** (*cf.* def. 2) of any regulated regime with respect to the range of unregulated regimes to which the regional fauna may be adapted. The approach presented here provides a tool to identify (and if needed rank) strongly anomalous flow regimes on dammed systems. It is assumed here that, in many cases, identifying anomalous regime at hydro sites (compared to regional norms) is simpler and possibly more relevant to managers than quantifying the precise degree of regime alteration, pre- versus post- damming. For example, when an altered flow regime due to hydro regulation remains well within the range of regimes

observed in comparable unregulated rivers with healthy fish populations, the potential for negative effects on similar fish populations at the hydro site may be assumed to lower.

Note however that analyses of regime alteration (comparing pre- versus post- damming periods) are still useful and in some contexts may even be required to understand damming impacts on some key habitat types. Two examples of possible pathways of effects on fish populations of modest flow regime alterations are given next: these involve possibly significant changes to bed sediment composition in some reaches triggered by modest changes to high flows, or changes to summer thermal regimes triggered by modest changes in low flow regimes.

The details of natural bed sediment transport patterns and local grain size composition along alluvial rivers are well known to be highly sensitive to the particular high flow regime of each river. Sand, gravel, cobble bed composition and transport levels can be strongly modified after only minor reductions in mean annual flood levels and in ways that can affect fish habitat. In particular, in geomorphologically sensitive reaches along the newly regulated system (for example in non-bedrock reaches located close downstream of unregulated tributaries carrying high sand or silt loads), even a minor alteration of the regulated mainstem flow regime can have significant long-term impacts on post regulation wetted channel widths and bed sediment composition. The resultant long-term geomorphic changes to such sensitive reaches often take decades to become manifest, usually as an increase of sand encroachment within and/or the eventual burial of prior river bed gravels. These changes can significantly affect key fish spawning or invertebrate prey habitats. Interesting examples of various impacts to the geomorphic structure of fish habitats following regulation are given in Ligon et al (1995). However, because of the sensitivity of bed sediment transport patterns to flow levels, in some cases minor alterations in mean flood levels that can have significant impacts on fish habitats in some reaches of the regulated river may not be large enough to show up as a clear, regional scale anomaly in flood levels. Thus precise analyses of flow regime alteration (pre versus post), focusing on mean annual flood metrics, may remain important to predict such effects.

Similarly, estimates of degree of **alteration** (pre vs. post) of extreme, summer low flow levels (e.g. the 7day duration low flow of a few year recurrence, etc.) may be needed to predict regulation effects on summertime wetted habitat area, water temperatures and associated thermal stress levels affecting fish populations, particularly at the southern limit of their range. Here again, regional anomaly estimates of such low flow metrics may not be sensitive enough to detect regime alterations with significant effects on summer thermal stresses in some reaches.

LITERATURE CITED

Acreman, M. C., & Dunbar, M. J. 2004. Defining Environmental River Flow Requirements – a review. Hydrology and Earth System Sciences, 8(5), 861-876.

Arthington, A. H., Bunn, S. E., Poff, N. L., & Naiman, R. J. 2006. The Challenge of Providing Environmental Flow Rules to Sustain River Ecosystems. Ecological Applications, 16(4), 1311-1318.

Boisclair, D., Lapointe, M., Saint-Hilaire, A., Rasmussen, J.B., Seney, C., Lanthier, G., Bourque, G., Guénard, G., Macnaughton, C.J., Harvey-Lavoie, S. 2016. Modelling the effects of chemical and physical drivers on fisheries productivity metrics across rivers of varying hydrological regimes: lessons learned from NSERC HydroNet 2009-2015. DFO Can. Sci. Advis. Sec. Res. Doc. 2016/033. viii + 62 p.

Caissie, D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51: 1389–1406.

Church, M. 2002. Geomorphic Thresholds in Riverine Landscapes. Freshwater Biology, 47(4), 541-557.

Daigle, A., St-Hilaire, A., Beveridge, D., Caissie, D., & Benyahya, L. 2011. Multivariate Analysis of the Low-Flow Regimes in Eastern Canadian Rivers. Hydrological Sciences Journal, 56(1), 51-67.

Fausch, K. D., Taniguchi, Y., Nakano, S., Grossman, G. D., & Townsend, C. R. 2001. Flood Disturbance Regimes Influence Rainbow Trout Invasion Success Among Five Holarctic Regions. Ecological Applications, 11(5), 1438-1455.

Junk, W. J., Bayley, P. B., & Sparks, R. E. 1989. The Flood Pulse Concept in River-Floodplain Systems. Canadian Special Publication of Fisheries and Aquatic Sciences, 106, 110-127.

Kennard, M. J., Pusey, B. J., Olden, J. D., Mackay, S. J., Stein, J. L., & Marsh, N. 2010. Classification of natural flow regimes in Australia to support environmental flow management. Freshwater Biology, 55(1), 171-193.

Ligon, F. K., Dietrich, W. E., & Trush, W. J. 1995. Downstream Ecological Effects of Dams: A geomorphic perspective. BioScience, 45(3), 183-192.

Magilligan, F. J., & Nislow, K. H. 2005. Changes in hydrologic regime by dams. Geomorphology, 71, 61- 78.

McLaughlin, F., Laporte, M., Bourque, G., Boxclair, D. 2014. Using regional flow classes as references to analyse flow regime anomalies across a set of Canadian rivers regulated for hydro-electric production. Journal of Hydrology, 519: 307-28

McLaughlin, Fraser W.R. 2014. McGill University. (available online). Using Regional Flow Regime Classes to Identify Flow Anomalies in a Set of Canadian Rivers Regulated by Dams.

McManamay, R. A., Orth, D. J., & Dolloff, C. A. 2012a. Revisiting the Homogenization of Dammed Rivers in the Southeastern US. Journal of Hydrology, 424-425, 217-237.

McManamay, R. A., Orth, D. J., Dolloff, C. A., & Frimpong, E. A. 2012b. A Regional Classification of Unregulated Stream Flows: Spatial Resolution and Hierarchical Frameworks. River Research and Applications, 28, 1019-1033.

McManamay, R. A., Orth, D. J., Dolloff, C. A., & Frimpong, E. A. 2012c. Regional Frameworks Applied to Hydrology: Can Landscape-Based Frameworks Capture the Hydrologic Variability? River Research and Applications, 28, 1325-1339.

Mims, M. C., & Olden, J. D. 2012. Life History Theory Predicts Fish Assemblage Response to Hydrologic Regimes. Ecology, 93(1), 35-45.

Monk, W. a., Peters, D. L., Allen Curry, R., & Baird, D. J. 2011. Quantifying Trends in Indicator Hydroecological Variables for Regime-Based Groups of Canadian Rivers. Hydrological Processes, 25, 3086 - 3100.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E. 1997. The Natural Flow Regime. BioScience, 47(11), 769-784.

Poff, N. L., Olden, J. D., Merritt, D. M., & Pepin, D. M. 2007. Homogenization of Regional River Dynamics by Dams and Global Biodiversity Implications. Proceedings of the National Academy of Sciences of the United States of America, 104(14), 5732-5737.

Poff, N. L., Olden, J. D., Pepin, D. M., & Bledsoe, B. P. 2006. Placing Global Stream Flow Variability in Geographic and Geomorphic Contexts. River Research and Applications, 22, 149-166.

Poff, N. L. 1996. A Hydrogeography of Unregulated Streams in the United States and an Examination of Scale-Dependence in some Hydrological Descriptors. Freshwater Biology, 36, 71-79.

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., Acreman, M. 2010. The Ecological Limits of Hydrologic Alteration (ELOHA): A New Framework for Developing Regional Environmental Flow Standards. Freshwater Biology, 55, 147-170.

Poff, N. L., & Zimmerman, J. K. H. 2010. Ecological Responses to Altered Flow Regimes: A Literature Review to Inform the Science and Management of Environmental Flows. Freshwater Biology, 55(1), 194-205.

Richter, B. D., Baumgartner, J. V., Powell, J., & Braun, D. P. 1996. A Method for Assessing Hydrologic Alteration within Ecosystems. Conservation Biology, 10(4), 1163-1174.

Richter, B. D., Baumgartner, J., Wigington, R., & Braun, D. P. 1997. How Much Water does a River Need? Freshwater Biology, 37, 231-249.

Smokorowski, K. E., Bergeron, N., Bowman, D., Clarke, K., Cooke, S., Curtis, R., Dawson, J. 2011. NSERC 's Hydrofate: A National Research Network to Promote Sustainable Hydropower and Healthy Aquatic Ecosystems. Fisheries, 36(10), 490-495.

Steuer, J. J., Stenvold, K. a., & Gregory, M. B. 2010. Determination of Biologically Significant Hydrologic Condition Metrics in Urbanizing Watersheds: An Empirical Analysis over a Range of Environmental Settings. Hydrobiologia, 654, 27-55.

Stromberg, J. C., Beauchamp, V. B., Dixon, M. D., Lite, S. J., & Paradzick, C. 2007. Importance of Low-Flow and High-Flow Characteristics to Restoration of Riparian Vegetation along Rivers in Arid South-Western United States. Freshwater Biology, 52(4), 651-679.

Whitfield, P. H., & Cannon, A. J. 2000. Recent Variations in Climate and Hydrology in Canada. Canadian Water Resources Journal, 25(1), 19-65.

Williams, J. T., Parker, M. S., & Power, M. E. 1988. Effects of Disturbance on River Food Webs. Science, 273(5281), 1559-1561.

Zimmerman, J. K. H., Letcher, B. H., Nislow, K. H., Lutz, K. A., & Magilligan, F. J. 2010. Determining the Effects of Dams on Subdaily Variation in River Flows at a Whole-Basin Scale. River Research and Applications, 26, 1246-1260.

SECTION 3 - CHEMICAL DRIVERS: Nutrients as chemical drivers of fish biomass: empirical models of the relationship between the nutrient regime and fish biomass, and the effect of impoundment on river nutrient regimes

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PART I: Modelling fish biomass as a function of nutrient richness

Fish biomass (FB) can be used as an estimate of the productivity of an ecosystem, and the relationship between the fish biomass and the concentration of chemical nutrients in the water can be used to evaluate the importance of nutrients as drivers of fish productivity. Since nutrients are well known to drive both primary productivity and primary producers in rivers (Oglesby and Allen 1974, Bachmann and Jones 1974) and lakes (Dillon and Rigler 1974, Peterson et al. 1993) it is reasonable to expect that energy flow from primary producers to the "food base" that supports fish secondary productivity, would be similarly dependent on the nutrient regime (Oviatt and Peterson 1982, Hyatt et al. 2004).

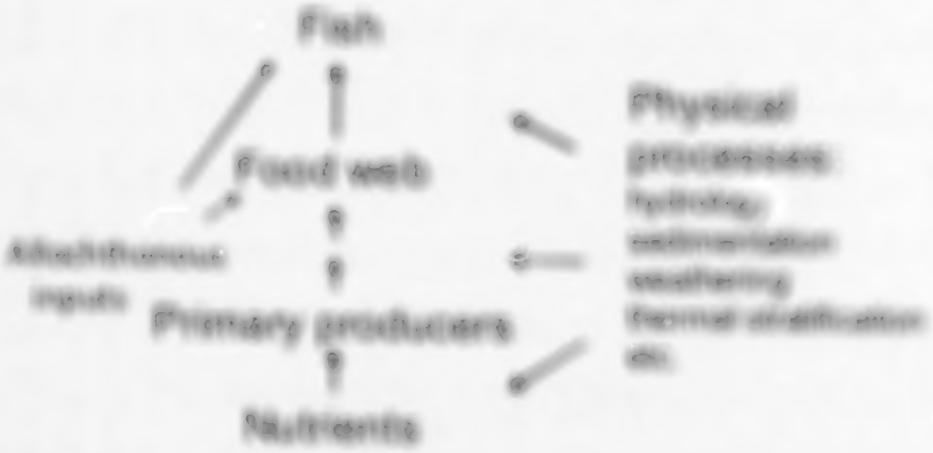

Figure 61. The dependence of fish abundance and productivity on the underlying trophic regime, and ultimately the nutrient regime, and the impact of physical factors that shape this habitat.

Physical factors can modify the pathways through which nutrients cycle and energy flow, thus hydrology (flow regime, thermal stratification, wave action, etc.) and the climatic regime can influence the rates of all chemical and biological processes. Such factors can shape the nutrient and thus affect fish directly, or indirectly, via their effect on rates and inter-specific interactions among fish, and on the distribution and removal rate of food.

Effects of the nutrient regime can arise through geographic, watershed or geomorphological differences in nutrient export capacity of watersheds, through factors that affect the rate and distribution of nutrients within the drainage network, or through anthropogenic inputs (i.e. eutrophication) or land use differences. Regardless of the source of nutrient variability, trophic status is among the most pervasive contributors to system variability, and needs to be factored into any comparisons of productivity, no matter what the ongoing level of interest.

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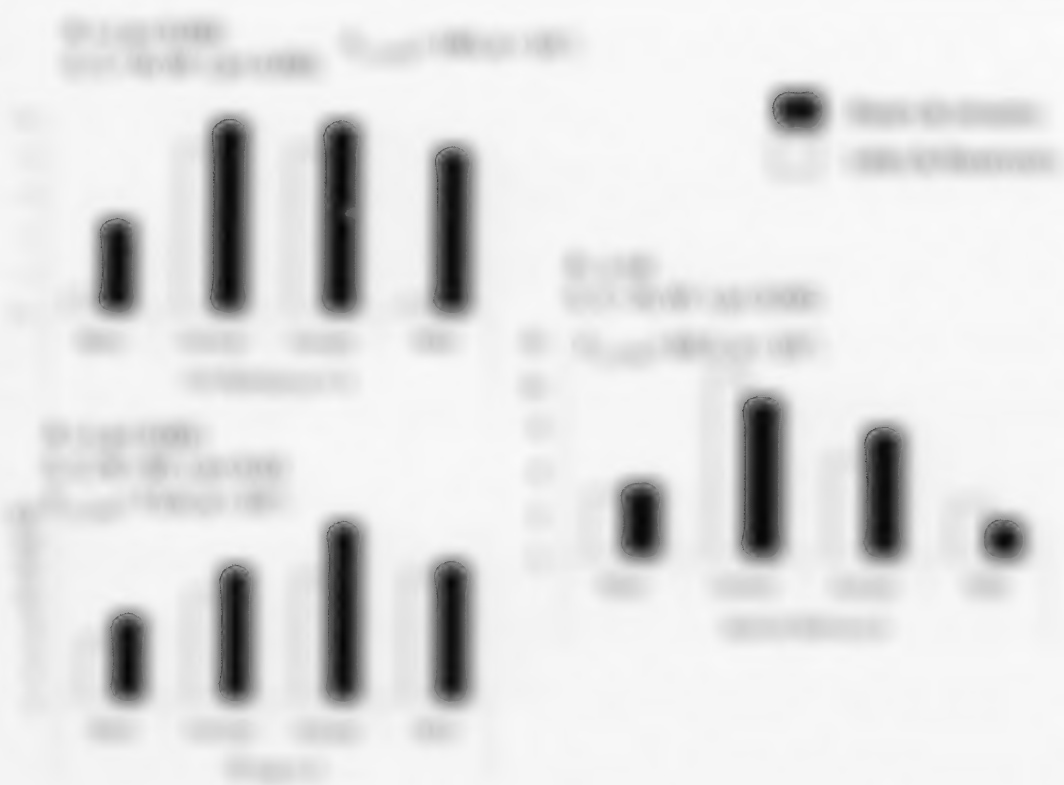


Figure 4B. Summary of the effects of [illegible] on [illegible] in the presence of [illegible] and [illegible] in the presence of [illegible] and [illegible] in the presence of [illegible] and

lake's productive capacity. Since the regional fish faunas are so species poor due to zoogeographical barriers that have limited post-glacial access from major refugia (Van Zyll de Jong et al. 2005; McPhail and Lindsey 1970), there are rarely other species present that are capable of compensating for such recruitment limitations, and as a result total fish biomass of such systems can be far below what the lakes could support. Recruitment limitations of this kind are not limited to lakes, but can of course occur in rivers and streams as well especially upstream of waterfalls or other barriers, but the generally greater availability of salmonid spawning and habitat, and the overall higher connectivity of rivers, makes rivers more likely to approach their productive capacity than lakes in species poor regions.

Recruitment limitation limits lake fish biomass in differently in these two species poor regions. The lakes in the Cordillera are much larger and deeper, their watersheds are steeper, and their oligotrophy is much more extreme than in Newfoundland. Moreover, the salmonids that use the lakes are different, and the role of lakes in their life history is also different. In the West, the main species utilizing the large, deep fjord lakes is the Sockeye Salmon (*Oncorhynchus nerka*) and their land-locked counterpart the Kokanee salmon. Both spawn primarily in streams, usually inflows, and migrate to the lakes as fry, and spend one or more years as pelagic zooplanktivores, before migrating out, either as smolts going to sea for one or more years, or as adults migrating to spawn.

Hydroacoustic fish surveys were found for 66 such 'sockeye' lakes, and of these, only 16 were considered to have fish communities that approached productive capacity as judged by primary productivity, zooplankton biomass and nutrient richness (Shortreed et al. 2001, Hyatt et al. 2004). The remainder often had fish populations as much as 10-fold lower than the trophic regime was estimated to be able to support, and were considered unlikely to respond to nutrient fertilizations. For these lakes FB was uncorrelated with TP, and the mean size of fry or outmigrating smolts was uncorrelated to abundance (Hyatt and Stockner 1985). The most common reason cited for sockeye and kokanee populations being below capacity was limited access to good spawning habitat, with migration barriers also an often cited factor (Shortreed et al. 2001). Restricted access to suitable spawning gravels is greatly exacerbated by the steep terrain leading to washout of fine gravels, and waterfalls and impassible 'chutes' near the lakes on most inflow streams. In only one case was the presence of another species, the Three-Spined Stickleback *Gasterosteus aculeatus*, listed as a factor limiting sockeye abundance (Shortreed et al. 2001). While there are some other salmonids that do sometimes use these lakes, none are efficient pelagic zooplanktivores or lake spawners (McPhail 2007). As a result, poor species richness in the regional fauna of the Cordilleran region of Western North America makes the probability of 'under saturation' due to recruitment limitation greater, and this factor has its greatest impact on fish biomass in the large deep fjord lakes.

The lakes of Newfoundland are much shallower and less oligotrophic than the western fjord lakes, and while the geography of the landscape is much less steep, fine gravel can still be limiting in some areas because most glacial outwash is below sea level due to post glacial subsidence and recession in marine sea level (Batterson and Liverman 2010). Many of the lakes appear as widenings along relatively low gradient rivers to which riverine fish have ready access (Colbo et al. 2011).

The main fishes that use the lakes are Atlantic Salmon parr (*Salmo salar*) and juvenile Brook Trout (*Salvelinus fontinalis*) (Clarke and Scruton 1999). Both are generalist feeders, utilizing mainly littoral benthic macroinvertebrates and neither are effective zooplanktivores. The Arctic Charr (*Salvelinus alpinus*) is present in Newfoundland, and its juvenile stages are efficient zooplanktivores, but the species is found in very few of the lakes (Van Zyll de Jong et al. 2005). The Three-Spined Stickleback is also found in some of the lakes but its contribution biomass is small (Ryan 1986). Newfoundland Atlantic Salmon are atypical in that they often move from

rivers into lakes and feed on littoral macroinvertebrates (Ryan 1986). This likely reflects the fact that Newfoundland lakes lack the diverse array littoral zone competitors found in most continental lakes (Ekinaro and Gibson 1999). Similarly, the Brook Trout found in Newfoundland lakes are also juveniles that have moved from stream nursery habitats to lakes as larger juveniles. Neither species spends more than a fraction of its lifecycle in the lakes; the salmon smolt and migrate to sea following their lacustrine phase, and the brook trout move to streams and spawn.

The extremely low biomass in many of the Newfoundland lakes relative to their associated rivers and to their nutrient levels, is an indication of under-saturation. Moreover, the biomass of fish in these lakes is uncorrelated to nutrient richness, despite considerable range in trophic richness (Cote et al. 2011) and size of parr is not correlated with density (Ryan 1986). The abundance of both salmonid species within river systems is likely determined by the capacity of the spawning and nursery habitat in the streams and rivers, and the abundance of fish in the larger lakes appears to bear little relation to what the lake resources could support, since the lacustrine phase of the salmon and Brook Trout in Newfoundland is only a minor portion of their overall life-history. Thus the recruitment limitation leading to under-saturation in Newfoundland lakes likely results from lake habitat being available in higher proportion than spawning and rearing habitats for the two main salmonids that use the river systems. Just as on the west coast, the poor species richness in the Newfoundland fish fauna makes the probability of 'under saturation' due to recruitment limitation greater, since there is a lack of lacustrine specialists, especially zooplanktivores, whose abundance would be limited by the lake resources rather than the recruitment capacity of associated rivers and streams.

Interestingly, species-poor systems are not confined to salmonid dominated coastal regions. In Central North America and Europe, there are species poor headwater systems where the one or two species present are either Yellow Perch (*Perca flavescens* or *P. fluviatilis*), cyprinids (often the Roach *Rutilus rutilus*, but Northern Red-Belly Dace and Golden Shiner lakes are also found), centrarchids (usually Bluegill or Pumpkinseed Sunfish) or catostomids (usually White Sucker) (Horppila et al. 2010; Kelso and Johnson 1991).

While biomass in these lakes is invariably low compared to similar multi-species lakes where these same species are found in communities of 5-10 species, there are no cases of extreme under production, where biomass is over an order of magnitude deficient relative to trophic resources, such as the cases described in coastal salmonid lakes. The likely reason for this is that recruitment limitation is much less likely to occur in perch, minnow or sucker lakes, since these species are capable of spawning and completing their life cycles in lakes or slow moving streams, and do not have specialized requirements such as clean fine gravels. As a result bottlenecks related to access to spawning grounds and/or habitat fragmentation are not likely to occur in species poor perch, minnow or sucker systems.

*Table B1. Comparison of mean values (arithmetic mean ± SE, geometric mean ×,÷ error factor, and median and 25th & 75th %iles) of FB, TP, Sp and N within and among regions and ecosystem types. Geometric mean = 10^mean log10, and error factor (%) = [1 - 10^(SE of log10 values) * 100].*

	Fish Biomass (FB) $g\,m^{-2}$	Total Phosphorus (TP) $mg\,m^{-3}$	Total Nitrogen (TN) $g\,m^{-3}$	Species Richness (#Sp)	N
General	7.57 ± 3.9 3.47, 9.6% 3.88, 1.23, 9.5	30.0 ±1 3.0 16.3, 33% 15, 6.1,27.6	0.59 ± 0.12 0.28, 13% 0.21,0.12, 0.56	6.7 ± 3.5 3.72, 32% 3, 2, 8	235
Rivers and streams	10.2 ± 2.8 5.03, 28% 5.5, 2.4, 12.5	41.9 ± 11.2 22.4, 20% 22.5, 10.5,110	0.82 ± 0.16 0.51, 18% 0.43,0.22, 1.90	6.2 ± 3.4 3.9, 16% 3, 2, 8	84
Lakes and Reservoirs	4.9 ± 2.1 1.4, 31% 1.7, 0.6, 5.5	13.8 ± 1.6 8.9, 10% 7.9, 5.0, 20	0.22 ± 0.05 0.19, 13% 0.15,0.10, 0.42	7.1 ± 2.5 3.7, 15% 3, 2, 8	151
Newfoundland Labrador	4.9 ± 2.5; 2.2, 19% 2.1, 0.8, 3.7	30.3 ± 6.0; 24.2, 15% 22, 10.7,32.5	0.48 ± 0.10; 0.30, 23% 0.30,0.20, 0.54	2.1 ± 0.09; 1.9, 20% 2, 1, 3	34
Central Canada and USA	12.1±2.5; 6.9±24% 5.1,2.1, 12.5	22.9±6.5; 16.4±16% 13, 7.5,21.5	0.35±0.10; 0.26±21% 0.27,0.14, 0.45	15.5 ±1.5; 9.5±13% 8, 5, 12	78
Western Cordillera	1.61±0.6; 0.62±25% 0.61,0.39,1.08	6.62±1.25; 5.21±13% 5.1, 3.2,7.0	0.12±0.0.02; 0.0.07±18% 0.10,0.06, 0.13	.9 ±0.7; 2.4±16% 2, 2, 4	95
Europe	11.7 ± 2.8 7.4, 16% 8.5,4.1, 17.2	60.1 ± 11.3 35.4, 20% 49, 29.5,105	1.31 ± 0.25 0.85, 22% 0.82,0.35, 1.92	6.3 ± 1.2 4.3, 16% 5, 3, 10	28

The relationship between trophic richness and fish biomass

As is the case for other components of aquatic ecosystems such as phytoplankton (Dillon and Rigler 1974; Bachmann and Jones 1974) and zooplankton, and zoobenthos (Hanson and Peters 1984; Rasmussen 1988), by far the best statistical measure of nutrient richness for assessing the trophic richness of systems for fish is the total phosphorus (TP) concentration of the water (Hanson and Leggett 1982; Ney et al. 1990; Randall et al. 1995)—as predicted by Hypothesis 1. While total nitrogen (TN) is generally correlated with total phosphorus, and can sometimes provide reasonable predictions of trophic richness, TN is usually present in excess of the Redfield ratio, and more variable than TP, and as a result is a much weaker predictor, except in cases of severe N-limitation (Dillon and Rigler 1974).

Since nutrient-based models of fish productive capacity are intended to estimate the potential biomass that a given trophic regime can support, systems that exhibited 'under-saturation' due to recruitment limitation, as outlined previously were excluded from the regression analyses. Thus of the 235 estimates listed in Table B1, 201 (84 rivers, 117 lakes) from the four geographic regions were included in the log-log regressions (Tables B2-B4).

Log TP was a highly significant predictor of Log FB in all regions; moreover rivers and lakes did not differ in these FB vs TP models indicating that the main driver of higher fish biomass in rivers (Table B1) was their overall greater nutrient richness.

Although Log TP was a significant predictor in all regions, both intercept and slope varied significantly among regions; intercepts (Table B2) varied by more than an order of magnitude (-0.83 to 0.24), and slopes also varied significantly from 0.63 to 1.26. Intercept and slope estimates were not correlated to each other, but intercepts were highly correlated to mean species richness of the data set (r=0.95, n=6, p=0.003), with the highest value in the species rich Appalachian reservoirs, and lowest values in the West and Newfoundland. Intercepts were also negatively correlated with the precision of the model (R^2), with the strongest regressions tending to have higher intercepts (r=0.69, n=6, p=0.05). While slope was not significantly correlated with either R^2 or species richness, the data sets with the greatest range of species richness tended to have the lowest R^2 values.

The close relationship between intercept and species richness implied that much of the residual variability in these FB vs. TP relationships is related to species richness, and when Log S was tested it was highly significant in the overall model (Figure B4) and in all but the smallest of the regional data sets (Table B3). Thus even in regions such as Newfoundland and the West, where species richness was low and did not vary over a broad range, it still made a statistically significant contribution to the models. Besides the overall model, Log S made the strongest contribution in the Central region, where species richness was overall highest, and also exhibited the greatest range of variation (Table B1). Predictions were also improved slightly by the addition of Log mean depth, which was a significant (-) factor in the overall model and in all but one of the regions (Table B3).

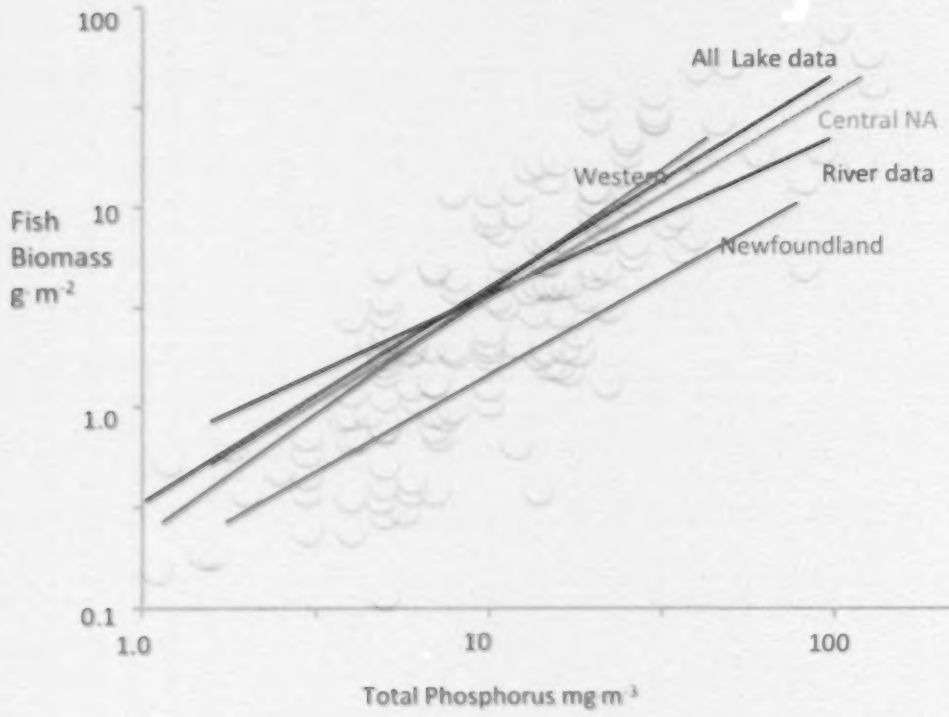


Figure B3. Log FB vs Log TP Relationships for different regions, compared to previously published relationships.

These multiple regression models retain the predictive effect of log TP, which was strong in all regions and no longer significantly variable among regions. They incorporate the predictive power of species richness, which captured all of the significant regional differences of the previous model, and the effect of mean depth (Log Dm t=-3.0). This variable captured both the difference between shallow and deep lakes, and also helped capture the effect of flowing versus standing water. While rivers, in general, had higher fish biomass than lakes (Table B1), they also had higher TP concentrations on average than lakes, and thus the predictive power of TP, combined with mean depth captures the main differences both within and among lakes and rivers, and the regional differences as well, to the extent no region had significant residuals, and in addition there were no significant interaction terms between region and the slope of either LogTP, Log S or Log Dm.

Table B2. Comparison of Intercepts and Slopes among regions, and with previously published studies.

Log_{10} FB = Intercept + slope * Log_{10} TP	Intercept ± SE	Slope ± SE	RMS, R^2	Log S # species	N
Central NA	$-0.50_{\pm 0.13}$	$+1.11_{\pm 0.11}$	0.36, 0.56	1.18	77 lakes, rivers
Western NA	$-0.63_{\pm 0.10}$	$+0.91_{\pm 0.13}$	0.30, 0.44	0.46	64 lakes, rivers
Newfoundland	$-0.78_{\pm 0.30}$	$1.01_{\pm 0.23}$	0.41, 0.45	0.30	34 lakes, rivers
Europe	$-0.69_{\pm 0.24}$	$+0.95_{\pm 0.18}$	0.33, 0.69	0.80	26 lakes, rivers
All lake data	$-0.46_{\pm 0.07}$	$+1.14_{\pm 0.06}$	0.26, 0.59	0.42	151
All river data	$-0.21_{\pm 0.08}$	$+0.70_{\pm 0.07}$	0.36, 0.45	0.45	78
Hanson & Leggett 1982	-0.23	0.71	MSE =?, 0.75	---	18 Central lakes
Ney et al. 1990	0.24	1.02	MSE =?, 0.84	1.70	N=21 Appalachian Reservoirs
Randall et al. 1995	$-0.29_{\pm 0.16}$	$0.63_{\pm 0.11}$	0.28, 0.65	---	18 Global lakes & rivers

Why is species richness such a strong predictor?

The power of fish species richness to influence the relationship between trophic richness and fish biomass has been little studied, and in fact, has been generally ignored by aquatic ecologists. Since species differ in life history, habitat requirements, and sensitivity to environmental risks, species richness can potentially buffer systems against recruitment limitation, thus providing redundancy by hedging against risks. However, this is not likely an important factor here since the sites that were considered to be strongly undersaturated were removed *a priori* from the data set. The contribution of species richness in the multiple regression models (Table B3) likely reflects functional diversity among species in trophic ecology. Thus the differences among species in food resources that they use, and the associated food web pathways and production sources that drive energy flow within the community likely means communities with greater species richness have access to a broader range of energy and nutrient sources (Carey and Wahl 2011). Although most species overlap

trophically to some extent, and many are 'trophic generalists,' the extent to which they differ can lead to additive effects on community biomass. For example, the food chain in a lake that lacks pelagic zooplanktivores would make far less efficient use of plankton productivity than one that contained efficient zooplanktivores, and fish biomass, in the latter system should be higher. Similarly, efficient microbenthivores, herbivores, detritivores, deposit feeders, and periphyton feeders should all expand the trophic spectrum of the community significantly. This principle, the combination of redundancy and functional complementarity, has been shown to explain the observed increases in the productivity, biomass and stability of grasslands (Tilman and Downing 1994; Hooper et al 2005) and other plant communities; however, the significance of fish biodiversity for productivity has received little consideration.

Table B3. Comparison of multiple regressions predicting Log FB in different regions.

Log FB = A + B*Log TP+ C*logS+ D*log Dm	A ± SE	B ± SE	C ± SE	D ± SE	RMS, R^2	N
Central NA	$-.65_{\pm 0.11}$	$0.82_{\pm 0.10}$	$0.60_{\pm 0.08}$	$-0.21_{\pm 0.06}$	0.30, 0.74	77 lakes, rivers
Western NA	$-.59_{\pm 0.14}$	$0.72_{\pm 0.16}$	$0.37_{\pm 0.16}$	$-0.07_{\pm 0.03}$	0.28, 0.75	64 lakes, rivers
Newfoundland	$-.32_{\pm 0.21}$	$0.89_{\pm 0.16}$	$1.26_{\pm 0.33}$	$-52_{\pm 0.27}$	0.27, 0.76	34 lakes, rivers
Europe	$-.64_{\pm 0.23}$	$0.81_{\pm 0.16}$	$0.52_{\pm 0.32}$	$0.05_{\pm 0.20}$	0.30, 0.75	26 lakes, rivers
General Model	$-.89_{\pm 0.07}$	$1.03_{\pm 0.06}$	$0.59_{\pm 0.05}$	$-0.09_{\pm 0.03}$	0.28, 0.82	201 lakes, rivers

Species richness is likely only a crude metric by which to represent a potentially complex effect and further research may show that indices that quantify both redundancy and complementarity among species, will provide more powerful predictions. Log S is, however, a simple and reasonable proxy, whose predictive strength seems compelling (Table B3). The fact that species richness contributes almost as strongly to biomass prediction in some of the regional submodels as it does in the integrated data set that spans across region (Table B3) is potentially very important. This is so because species richness varies greatly across regions, and models that incorporate this variable might thus be prone to confounding regional differences. However, species richness also varies considerably within regions, as headwater lakes and streams, even in species rich areas such as Central Canada, often have very few species, due to dispersal barriers and other factors (Jackson et al. 2001). Thus although species richness varies considerably less within than across regions, the fact that it makes a strong contribution within regions, as well as among, weakens any argument that its effects simply result from spurious correlations with other regional differences. Moreover, the fact that adding Log S to the models succeeded in removing all significant residual regional differences between the regional submodels further supports its significance.

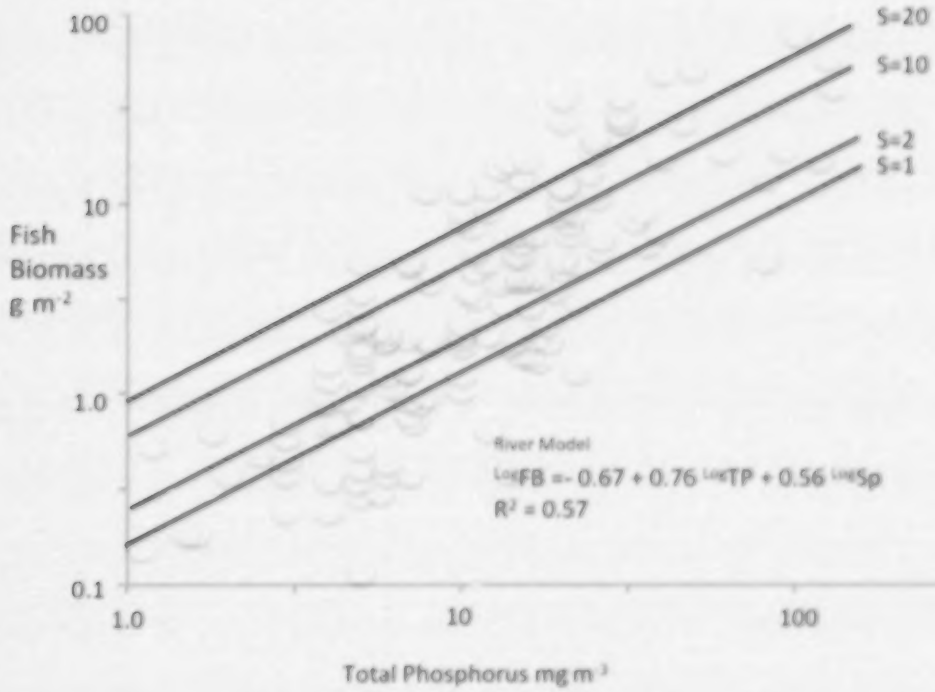


Figure B4. The combined influence fish of trophic richness (total phosphorus) and fish species richness on fish biomass. The plots are based upon the General model in Table B3 with mean depth set to 1m.

Lakes vs. Rivers

Although the previous analyses yielded models wherein rivers and lakes did not differ significantly, further analysis of the effect of depth for lakes showed that the slope of the Log FB vs Log TP relationship was increased significantly with lake depth. Thus shallow lakes had FB/TP slopes much less than 1, whereas the slope for deep lakes was close to 1.0. Thus the interaction between the two variables, Log TP* Log Dm was highly significant yielding the following multiple regression model for lakes and reservoirs.

$^{Log}FB =- 0.21_{\pm 0.09} + 0.34_{\pm 0.07}{}^{Log}TP + 0.68_{\pm 0.04}{}^{Log}Sp - 0.46_{\pm 0.07}{}^{Log}Dm + 0.38_{\pm 0.06}{}^{Log}Dm*{}^{Log}TP;$
$R^2 = 0.85$, RMS=0.26.

The interaction between TP and Dm is shown in Figure B5. Thus the effect of water column richness increases with depth of the water column, which likely reflects the increasing importance of water richness with increasing depth. The low FB vs TP slope for shallow lakes is reasonable given the potential for increased influences of benthic and allochthonous resources to productivity.

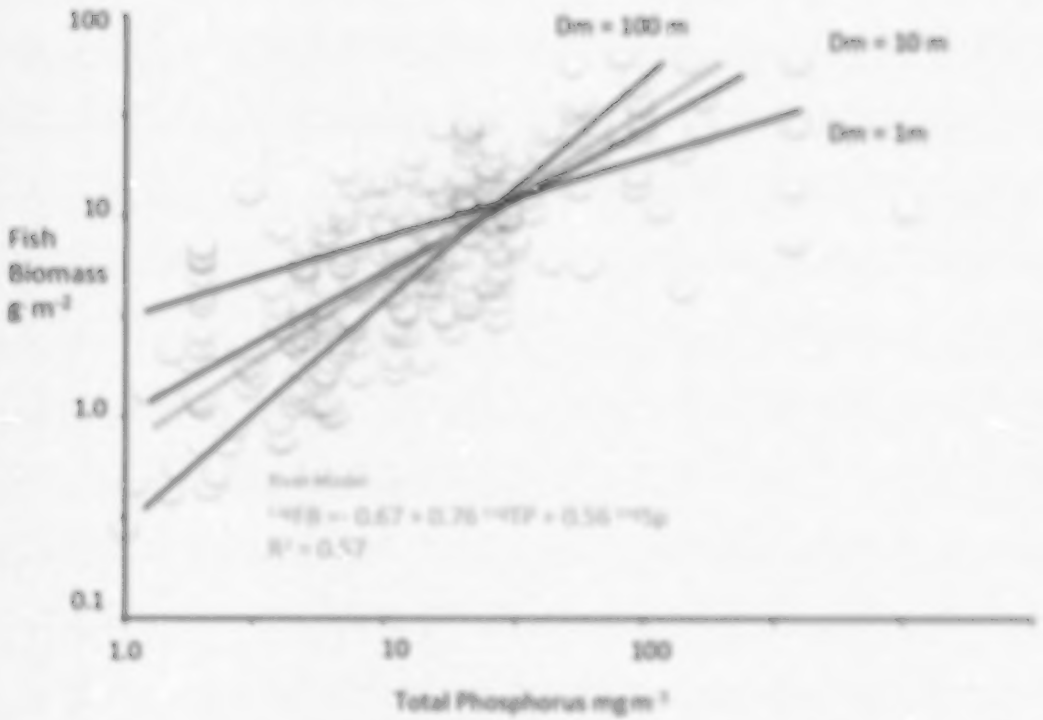


*Figure 85. The effect of lake depth on the relationship between Log FB and TP; a graphical representation of the TP * Dm interaction term in the multiple regression model shown. The general river model is shown for comparison (red).*

No interaction between TP and depth could be detected for rivers, and the slope of the FP vs TP relationship for rivers was intermediate between the values obtained for deep and shallow lakes.

$$^{Log}FB = -0.67_{\pm 0.09} + 0.76_{\pm 0.09}\ ^{Log}TP + 0.56_{\pm 0.06}\ ^{Log}Sp;\ R^2 = 0.57,\ RMS = 0.30$$

The coefficient for species richness was similar in magnitude and predictive power to that obtained in the lake model, but the overall precision of the river model was considerably lower than that obtained for lakes.

Using Production-Biomass models to estimate renewal rate of a fishery: a worked example based on Williston Reservoir

While TP has been used to predict commercial fish yields (Hanson and Leggett 1982) as well as sportfish yields (Jones and Hoyer 1982), the precision with which this can be done is limited, since the proportion of the total fish community that actually contributes to the fishery will depend on many factors, both ecological and social. The proportion of community biomass targeted by a fishery is usually very high in boreal lakes (Rawson 1951) whose communities contain whitefish, ciscoes, perch, walleye and lake trout of targetable size to commercial and recreational fisheries. While sockeye salmon smolts are not directly targeted by fisheries, they are of course a highly valued fisheries resource, and that can constitute as much as 90% of the biomass in many west coast lakes (Hyatt et al. 2004). Alternatively, many lakes may contain

significant biomass of large carp, chubs, shiners, bullheads and suckers (Carlander 1955) that are often not targeted by fisheries, and in such situations only a few percent of community biomass may be contributing to fisheries. Clearly, the proportion of the community biomass targeted by a fishery is highly variable, and will depend on the types of fish species present in a region, the structure of the food web, and to a very large extent on social factors such as which fish species are 'desirable' and which are not. None of these factors are direct or predictable functions of the nutrient regime, at least not on a broad scale across regions that differ broadly in their faunas and in their culture.

If however, from the fisheries management perspective a list of target species and their size ranges can be decided on and the proportion to which these fishes contribute to the total biomass of the community, this proportion can then be used as a weighting factor by which to multiply a community biomass prediction from models such as those described in Tables B3 & B4. The productivity/biomass (P/B) ratio (specific productivity) of these fish will be a function of climate/latitude and body size and can be predicted by models that relate P/B to latitude, mean annual temperature and body size (Randall 1982, Downing and Plante 1993), or from tabulated measurements of P/B for various species adjusted for latitude (e.g. Carlander 1969), and averaged across the species and sizes targeted by the fishery. These estimates of P/B can then be multiplied by the weighted biomass estimate to yield an estimate of the productive capacity of the fishery. Such an approach can either be conducted on a species by species basis, or averaged across the fishery.

To illustrate the use of P/B models to evaluate the productive capacity of a fishery, I will use a pertinent example of Williston Reservoir on the Peace River in north-eastern BC. Williston Reservoir has a mean TP value of 5.8 mg.m^{-3}, mean depth of 42 m, a fish community of 5 species, and its mean fish biomass based on hydroacoustic surveys is 1.6 g.m^{-2}. This mean biomass falls somewhat below the expected value from the general standing water model (1.8 g. m^{-2}).

$^{Log}P/B = -0.21_{ANCOVA} + 0.34_{ANCOVA}{}^{Log}TP + 0.68_{ANCOVA}{}^{Log}Sp - 0.46_{ANCOVA}{}^{Log}Dm + 0.36_{ANCOVA}{}^{Log}Dm^{2}{}^{Log}TP$;
$R^2 = 0.66$, RMS=0.26.

Since the RMS of this log-log model = 0.26, the error factor associated with this estimate is 1.82, and thus the fish biomass estimate for Williston Reservoir is well within the error bounds of the model.

The general river model predicts a fish biomass value of 2.0 g.m^{-2} for a river with TP 5.8 and 5 species. This log-log model (Figure B6)

$^{Log}P/B = -0.67_{ANCOVA} + 0.76_{ANCOVA}{}^{Log}TP + 0.56_{ANCOVA}{}^{Log}Sp$; $R^2 = 0.57$, RMS = 0.30

has an RMS value of 0.30 (error factor = 2.0), and thus the observed estimate is also well within the error bounds of this model, although the observation is consistent with the pattern that deep oligotrophic standing waters have fish biomass estimates that fall below rivers of the same trophic status (Figure B6).

The gill net samples taken in Williston Reservoir to compare with the hydroacoustic survey studies recorded 5 species of fish; however, lake whitefish were by far the most abundant, and constituted the only significant targetable species from a fisheries perspective. Lake Whitefish from 0.5 – 1.5 kg, (35-45 cm) was the range of targetable fish sizes in Williston Reservoir. Although this group made up > 85% of the biomass of fish from the gill net catches, the hydroacoustic analyses showed that this group in fact made up on average around 20% of the biomass of the Williston Reservoir fish community. This is consistent with other studies that have shown that gill net samples often overestimate the proportion of the community made up by large fish and underestimate the proportion made up by small fish (MacLellan and Hume

2011; Prchalova et al. 2008, 2009, 2010). While this may partly reflect mesh size biases, it mostly reflects the greater range of mobility of larger fish compared to small ones, which is a major contributing factor in determining the catch of gill nets and other passive gear (Prchalova et al. 2010).

In order to estimate the productive capacity of a fishery targeting the 0.5-1.5 kg whitefish which made up 20% of the total biomass of the reservoir, it is possible to first estimate the P/B ratio of these fish using a model of Downing and Plante (1993):

$$\log_{10} \text{Productivity (kg/ha)} = 0.20 + 0.83 \times \log_{10} \text{Biomass (kg/ha)} - 0.19 \times \log_{10} W(g) + 0.02 \times \text{MAT}(^{\circ}C)$$

For a fishery targeting a biomass of 3.2 kg.ha^{-1} (20% of total biomass of 1.6 g. m^{-2} or 16 kg.ha^{-1}), made up of fish maximum size 1.5 kg fish such as in the Witbank example (mean annual temperature, MAT = ..2°C), the model predicts productivity of 1.37 kg.ha^{-1}. This implies a P/B ratio of 1.37/3.2 of 0.33, which is somewhat lower than the values for lake whitefish in the Downing and Plante (1993) data set. Alternatively, the average of eighty lake whitefish P/B ratios for the populations in their data set was 3.21. Applying this value (i.e. 3.2 * 3.21) yields a productivity estimate of 0.67 kg.ha^{-1}.

If one then makes the assumption that a sustainable harvest should be no greater than 10 % of potential productivity (Downing and Plante 1993), then a reasonable estimate for the potential sustainable yield of such a whitefish fishery on Witbank Reservoir would be somewhere between 0.07 and 0.11 kg.ha^{-1}.

There are some potential issues with this approach that are difficult to assess, since the estimates of P and B upon which the Downing and Plante model was based include all age classes for a given species rather than just older larger sized individuals that would be targeted by the fishery. Thus specific productivity of these larger fish would be expected to be lower than that of the overall population. Part of the reason for the '10% assumption' mentioned above is to allow for the fact that the renewal rate of the biomass actually targeted by the fishery will be lower than that estimated for the whole population. Thus adherence to this fairly conservative rule of thumb should protect the fishery against overexploitation.

This worked example demonstrates that while fish biomass estimates are applicable to whole communities, and not just the portion of the community target by fisheries, fish biomass estimates such as those provided by biomass models can be used as the starting point to estimate the sustainable yield of a fishery.

PART 2: Effects of river impoundment/regulation on nutrient regimes and downstream fish biomass

In standing water, biological cycling of nutrients by plankton can produce both sinks, due to sedimentation (Stanley and Doyle 2002, Bosch 2008), and sources, due to internal loading (Nürnburg 2007). In standing water, impoundment of waterways by dams, therefore, has the potential to influence downstream nutrient regimes by disrupting the hydrological flow regimes, damping seasonal variability by trapping nutrient pulses (Power et al. 1996, Dunn and Arthington 2002), and in this way influence potential productivity for fish (Friedl and Wüest 2002, Puff and Allen 1980). Alternatively, a run-of-the-river facility may exhibit minimal upstream ponding, and thus have minimal downstream impact on nutrient regimes.

The extent to which the nutrient regimes of downstream receiving waters in regulated rivers are affected by impoundment would be expected to increase with biological productivity, the degree of thermal stratification and the extent to which down stream flows are hypolimnetic (Nürnburg 2007). Modified temperature regimes in stratified reservoirs can cause oxygen depletion in

[redacted]

Figure S4. Regional distribution of Canadian sampling sites. Triangle refers to location of multiple reference and regulated sites in each region. (R = S).

Table S4. Regulated and reference sites sampled by hydrofiber in different ecoprovinces of Canada.

Province	Area	Latitude	Longitude	Regulated and Reference	Ecoprovince
BC	West Kootenay R*	49.1158	-116.9458	R	Montane Cordillera
Ontario	Wagosh R.	48.6317	-89.4143	S	Boreal Shield
Ontario	Whitefish R*	46.6145	-84.1145	S	Boreal Shield
Quebec	Saint - Jean R*	48.1478	-71.2233	S	Boreal Shield
Quebec	Romaine R*	50.8873	-63.8888	S	Atlantic Maritime
Quebec	Puyjalon R*	51.5248	-66.1133	S	Atlantic Maritime
NB	Nas R.	47.1958	-64.1133	S	Atlantic Maritime
NB	Restigouche R*	48.0833	-67.1445	S	Atlantic Maritime
NS-Maritimes	Wolfville, Harbin, Lahave	44.4888	-113.1138	S	Montane Cordillera
BC	Mc R. (above Alta)	49.1178	-113.0888	S	Montane Cordillera
BC	Kicking Horse R. (above Wapta Lake)	49.5007	-116.4818	S	Montane Cordillera
Alberta	Ram R. (below Prairie Creek)	52.7205	-115.1168	S	Montane Cordillera
Alberta	Ram R. (above Ram Falls)	51.9345	-115.5208	S	Montane Cordillera
Alberta	Bighorn R. (above Abraham)	51.9358	-116.0842	S	Prairies
Alberta	Elk R.	49.5301	-114.0748	S	Prairies
Alberta	Waterton R.*	49.1111	-114.0833	S	Prairies
Alberta	St. Mary R.*	49.0813	-113.3833	S	Prairies
BC	Tsitika R.	50.1158	-126.7158	NR	Montane Cordillera
Ontario	Mississagua R*	46.9888	-84.5458	NR	Boreal Shield
Ontario	Ivanhoe R*	48.9845	-84.5138	NR	Boreal Shield
Quebec	Petite Saguenay R*	48.1417	-71.0478	NR	Boreal Shield

Figure 4.8 – Rivermouth, Mt. Lake Mill, and Mt. Lake Mill, and Mt. Lake Mill, and

Table 1.1: Vitamin C in samples of citrus fruit juices and blends from supermarkets/convenience shops/juice outlets

Name	Manufacturer	Total Cost(€)	Packaging	Type
Supermarket	Supermarket	0.99	1.00	Store own brand
Convenience	Convenience	0.75	1.00	Shop
Supermarket	Supermarket	0.99	1.00	Packaging
Juice	Juice	0.99	1.00	Store own brand
Blend	Blend, various sizes	0.75	1.00	Packaging
Supermarket	Supermarket	0.99	1.00	Packaging
M. Valley	M. Valley	0.75	1.00	Store own brand
Fruit Valley	Fruit Valley	0.99	0.75	Ingredient, not sugar
Valley	Valley*	0.99	1.00	Ingredient, not sugar
Supermarket	Supermarket	0.99	0.99	Store own brand
Blend	Blend	0.99	1.00	Packaging
Valleys	Valleys	0.99	0.99/0.75	Store own brand
Supermarket	Supermarket	0.99	1.00	Store own brand
Blend	Blend	0.99	1.00	Store own brand
Valley	Fruit juice co.	0.75	0.99/0.75	Store own brand
Valley concentrate	Valley concentrate	1.00	Valley Ltd.	Concentrate separately

C = Cryptic; SF = supermarket; SF = Convenience; C = Store own brand; Valley = Valley Limited

Results

Citrus concentrate manufacturers (C) have regulated many single portion different consumer regulated from undesignated. Citrus juice is an acceptable difference in manufacturers of concentrations. Vitamin C analysis. Vitamin C was in total. Vitamin C in C. Vitamin C is specifically different in the Vitamin C analysis. Vitamin C analysis C. Vitamin C in C. Vitamin C in C. Vitamin C in

regional models is used, and the upper boundary of 2 mg·m^{-3} predicts a range of 2.0 – 5.0 g m^{-2} (Figure B7).

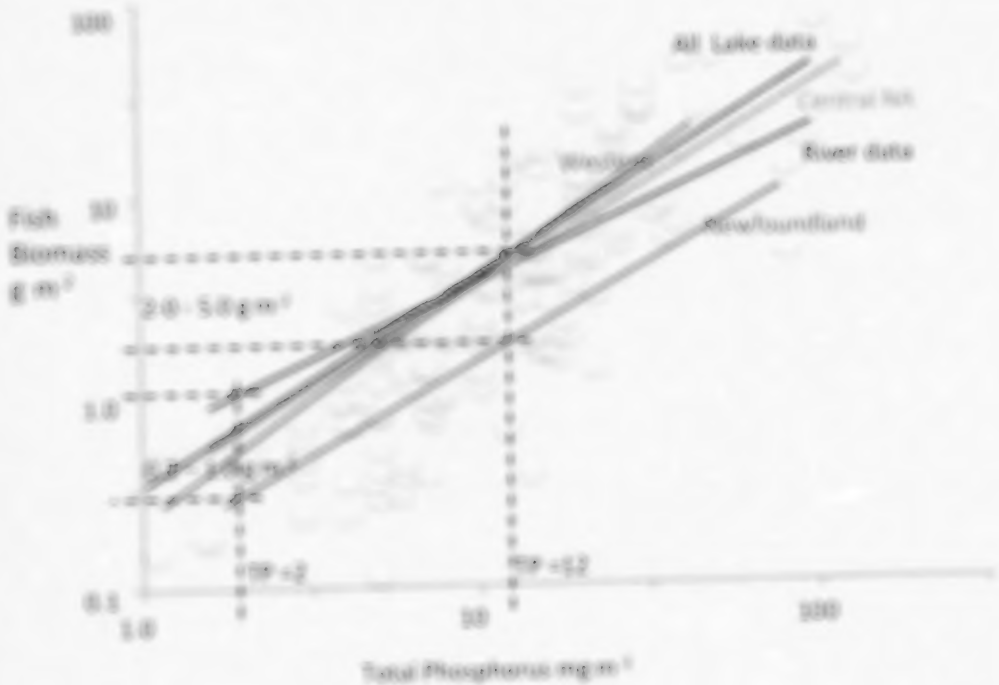


Figure B6. The range of Fish Biomass values predicted for a range of TP values from 2 - 12 mg m-3 for communities of 2, and 10 species, using the general multiple regression model shown in Table B3. Figure B4.

If values of species richness can be specified, then the models in Table B3, can be used, and the general model (Figure B4) would apply to all regions. For a fish community with 2 species, the TP range of 2-12 mg m^{-3} would predict a range of 0.4 -2.5 g m^{-2}, and for a fish community with 10 species, the TP range of 2-12 mg m^{-3} would predict a range of 1.1 -6.5 g m^{-2} (Figure B6). Thus the six-fold range of TP values found at the HydroNet sites across Canada, translate to approximately a six-fold range in fish biomass, but the magnitude of the values will depend on the species richness of the community in question.

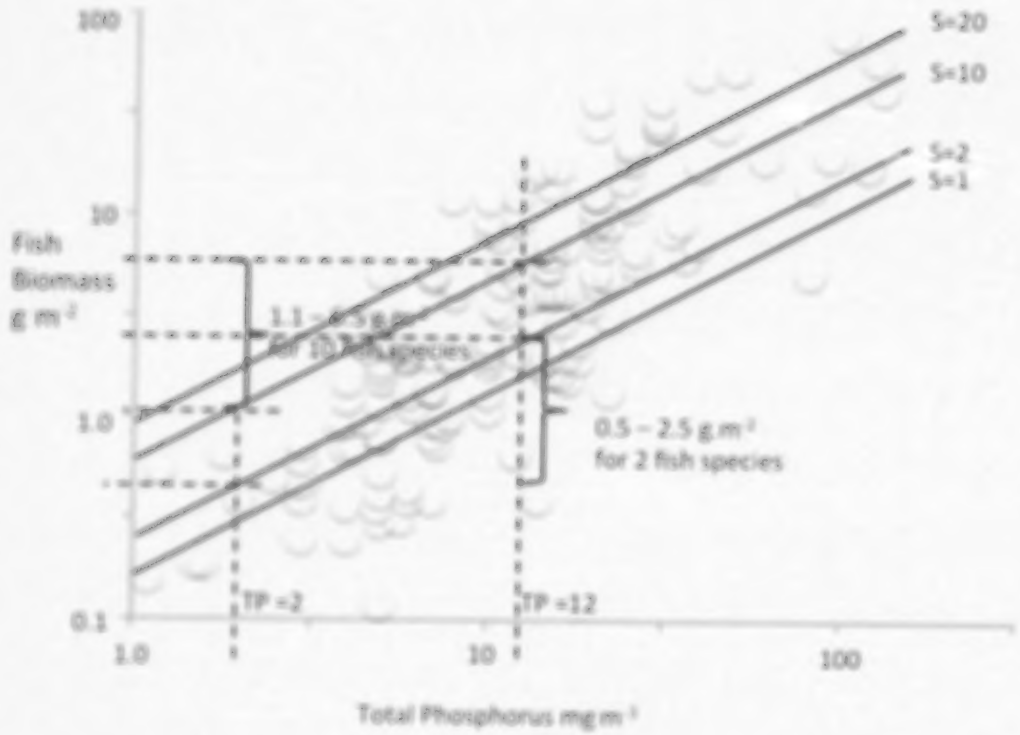


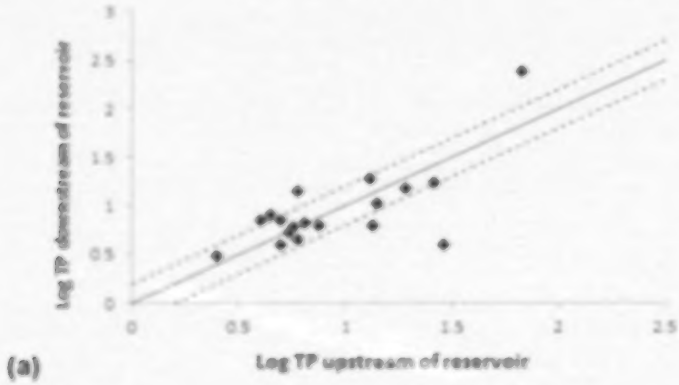
Figure B9. The predicted effect of the TP range found in Hydrofiet rivers for different levels of species richness, based on the river model shown in Figure B4.

Nutrient regimes upstream vs downstream of Reservoirs

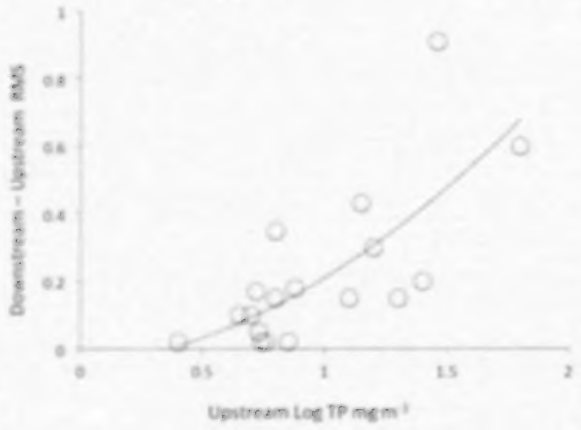
It was expected (Hypothesis II-2) that processes such as sedimentation and internal loading might lead to significant impoundment effects leading downstream differences in levels of N and P. However, most reservoirs had downstream TP values that were within 20 % of their upstream values and there was no significant directional bias to the upstream/downstream difference for either TP (paired $t_{15} = 1.74$, $p = 0.10$) or TN (paired $t_{15} = 0.82$, $p = 0.43$). Of the 4 reservoirs where large upstream/downstream differences in TP were found, two had higher values downstream indicating that internal loading within the reservoir was leading to downstream enrichment, and two had lower values, indicating that the reservoir was acting as a nutrient sink. The greatest nutrient sinks were Glenmore (run-of-the-river) and Spray Reservoirs (peaking). The two greatest nutrient sources were Twin Valley (irrigation storage) and Bearspaw (peaking). Although there was no directional bias, the % difference between downstream and upstream (root mean square difference) was an increasing function of TP (Figure B9) as hypothesized (II-2).

Thus it appears that whether the reservoir is a source or a sink, the larger the phosphorus load the greater the downstream effect is likely to be. Nutrients settle out in reservoirs as plankton develop and take up nutrients that enter from upstream sources (Finger et al. 2007), and oligotrophic reservoirs produce very low plankton crops (Friedl and Wuest 2002; Kennedy 2001). It is therefore reasonable to expect plankton sedimentation to trap a larger fraction of a

richer load. Internal loading of nutrients from accumulated sediment is more likely to occur in thermally stratified water bodies that experience hypoxia or anoxia in the bottom waters, a condition that is strongly associated with eutrophication (Nurnberg 2007).



(a)



(b)

Figure B10. TP sampling upstream and downstream of reservoirs in BC and Alberta. (a) Solid line indicates no change in TP up and downstream of reservoirs. Dashed lines indicate a 20% difference between up and downstream values, where the reservoir was acting as a sink (trapping) or source (releasing) nutrients downstream. (b) Root mean square (RMS) of downstream TP deviation relative to upstream TP; RMS = 0.18 Log TP2 + 0.088 Log TP -0.056 R2 = 0.53

Most of the BC and Alberta reservoirs in this study receive their runoff from mountain watersheds and are thus little influenced by agriculture in their watersheds. Prairie reservoirs situated along small streams with low discharge that receive little mountain runoff, can be greatly impacted by upstream agricultural inputs leading to cyanobacterial blooms (Mitchell and Prepas 1990). Twin Valley Reservoir situated along the Little Bow River is not only an example of this, but also a fairly young reservoir (built in 2004, Brinkmann and Rasmussen 2010). It experiences significant mid-summer *Aphanizomen* blooms that settle, decompose and deplete hypolimnetic oxygen leading to downstream enrichment. Thus nutrient levels along stream/reservoir watercourses will exhibit little longitudinal change if biological nutrient demand is minimal, however, when metabolism increases, both sedimentation and internal loading will increase making reservoirs greater sinks and/or sources, depending on their internal oxygen regimes.

The regulated hydroelectric rivers and their corresponding reference systems, although not free of agricultural influences are situated primarily in the boreal zone where such influences are minimal. Thus TP values were within the oligotrophic range (2 – 12 $mg.m^{-3}$, log TP = 0.3 -1.1) which places them at the low end of the trophic spectrum in Figure B9, and thus unlikely to experience significant longitudinal shift in nutrient concentration from upstream to downstream of reservoirs.

Fundamental/Scientific relevance

Establishing links between trophic richness, and the capacity of systems to support diverse and productive fish communities, and developing models that predict the standing crops that can be expected as a function of nutrient metrics and biodiversity are important goals for aquatic ecology and fisheries science.

The relationship between trophic richness and nutrient trapping and release to downstream sections of rivers by reservoirs is not well known, and to establish models that can predict longitudinal shifts in nutrient concentrations and productivity at all levels along regulated river courses is important for aquatic ecology and fisheries science.

Benefits to Industry and Government

Since nutrients are strongly related to productivity at all levels, it is important for industries, managers and regulators to have quantitative metrics of nutrient abundance and fish biomass. and good predictive models that can provide clear benchmarks of the amount of fish that a system can be expected to support, for a given level of richness, biodiversity and in different geographic regions.

Nutrient regimes differ within and among regions and assessing the impact of environmental disturbances on fish productive capacity will depend on understanding the baseline productivity that can be expected from a given type of community.

Literature Cited

Bachmann, R.W., Jones, J.R. 1974. Phosphorus inputs and algal blooms in lakes. Iowa State Journal of Research 49: 155-160.

Batterson, M., Liverman, D. 2010. Past and future sea level change in Newfoundland and Labrador: guidelines for policy and planning. Newfoundland and Labrador Dept of Natural Resources Geological Survey Report 10-1 pp 129-141.

Bosch, N.S. 2008. The influence of impoundments on riverine nutrient transport: An evaluation using the Soil and Water Assessment Tool. Journal of Hydrology 355: 131-147.

Bosch, N.S., Allan, J.D. 2008. The influence of impoundments on nutrient budgets in two catchments of Southeastern Michigan. Biogeochemistry 87:325-338.

Bosch, N.S., Johengen, T.H., Allan, J.D., Kling G.W. 2009. Nutrient fluxes across reaches and impoundments in two southeastern Michigan watersheds. Lake and Reservoir Management 25: 389-400.

Brinkmann, L., Rasmussen, J.B. 2010. High levels of mercury in biota of a new prairie irrigation reservoir with a simplified food web in Southern Alberta, Canada. Hydrobiologia 79: 393-402.

Bunn, S.E., Arthington, A.H. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management 30: 492-507.

Carey, M.P., Wahl, D.H. 2011. Determining the mechanism by which fish diversity influences production. Oecologia 167: 189-198.

Carlander, K.D. 1955. The standing crop of fish in lakes. Journal of the Fisheries Research Board of Canada. 12: 543-570

Carlander, K.D., 1969. Handbook of freshwater fishery biology, volume 1. The Iowa State University Press, Ames. Iowa. 752 p.

Chu, C., Minns, C.K., Mandrak, N.E. 2003. Comparative regional assessment of factors impacting freshwater fish biodiversity in Canada. Canadian Journal of Fisheries and Aquatic Sciences. 60: 624-634.

Clarke, K.D., Scruton, D.A. 1999. Brook trout production dynamics in the streams of a low fertility Newfoundland watershed. Transactions of the American Fisheries Society. 128:1222-1229.

Cote, D. 2007. Measurements of salmonid population performance in relation to habitat in eastern Newfoundland streams. Journal of Fish Biology. 70: 1134-1147.

Cote, D., Adams, B.K., Clarke, K.D., Langdon, M. 2011. Salmonid biomass and habitat relationships for small lakes. Environmental Biology of Fishes. 92: 351-360.

Deegan, L.A., Peterson, B.J. 1992. Whole river fertilization stimulates fish production in an Arctic tundra river. Canadian Journal of Fisheries and Aquatic Sciences. 49: 1890-1901.

Dillon, P.J., Rigler, F.H. 1974. The phosphorus-chlorophyll relationship in lakes. Limnology and Oceanography 19:767-763.

Downing, J.A., McCauley, E. 1992. The nitrogen-phosphorus relationship in lakes. Limnology and Oceanography. 37: 936-945.

Downing, J.A., Plante, C. 1993. Production of fish populations in lakes. Canadian Journal of Fisheries and Aquatic Sciences 50: 110-120.

Ekinaro, J., Gibson, R.J. 1999. Interhabitat migration of juvenile Atlantic salmon in a Newfoundland River system. Journal of Fish Biology. 51: 373-388.

Environment Canada. 2004. Canadian guidance framework for the management of phosphorus in freshwater ecosystems. Ottawa, Canada.

Finger, D.P., Bossard, P., Schmid, M., Jaun, L., Müller, B., Steiner, D. 2007. Effects of alpine hydropower operations on primary production in a downstream lake. Aquatic Sciences. 69: 240-256.

Friedl, G., Wuest, A. 2002. Disrupting biogeochemical cycles—consequences of damming. Aquatic Sciences 64: 55-65.

Gomez, E., Durillon, C., Rofes, G., Picot, B. 1999. Phosphate adsorption and release from sediments of brackish lagoons: pH, O_2 and loading influence. Water Research 33: 2437-2447.

Hanson JM, Leggett WC. 1982. Empirical prediction of fish biomass and yield. Canadian Journal of Fisheries and Aquatic Sciences 39:257-263.Hooper, D. U., Chapin, F. S., Ewel, J. J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J. H., Lodge, D. M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A. J., Vandermeer, J. and Wardle, D. A. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecological Monographs. 75: 3-35.

Horppila, J., Olin, M., Vinni, M., Estlander S, Nurminen L, Rask M, Ruuhijärvi J, Lehtonen H. 2010. Perch production in forest lakes: the contribution of abiotic and biotic factors. Ecology of Freshwater Fish 19: 257-266.

Hyatt, K.D., McQueen, D.J., Shortreed, K.S., Rankin, D.P. 2004. Sockeye salmon nursery lake fertilization: review and summary of results. Environmental Reviews 12: 133-162.

Hyatt, K.D., Stockner, J.G. 1985. Responses of sockeye salmon to fertilization of coastal lakes. Canadian Journal of Fisheries and Aquatic Sciences 42: 320-331.

Jackson, D.A., Peres-Neto, P.R., Olden, J.D. 2001. What controls who is where in freshwater fish communities—the role of biotic, abiotic and spatial factors. Canadian Journal of Fisheries and Aquatic Sciences 58: 157-170.

Jones, J.R., and Hoyer, M.V. 1982. Sportfish harvest predicted by summer chlorophyll a concentration in midwestern lakes and reservoirs. Trans. Am. Fish. Soc. 111: 176–179.

Kelso, J.R.M, Johnson, M.G. 1991. Factors related to the biomass and production of fish communities in small oligotrophic lakes vulnerable to acidification. Canadian Journal of Fisheries and Aquatic Sciences 48: 2523-2532.

Kennedy, R.H. 2001. Considerations for establishing nutrient criteria for reservoirs. Journal of Lake and Reservoir Management 17: 175-187.

MacLellan, S.G., Hume, J.M.B. 2011. Pelagic Fish Surveys of 20 Lakes in Northern British Columbia From 2006 to 2009. Canadian Technical Report of Fisheries and Aquatic Sciences 2950. 139 pp. Appendix 1-6.

McPhail, J.D., Lindsey, C.C. 1970. Freshwater Fished of Northwestern Canada and Alaska. Fish. Res. Board Can. Bull. 173, Ottawa.

Mandrak, N.E., Crossman, E.J. 1992. Postglacial dispersal of freshwater fishes into Ontario. Canadian Journal of Zoology 70: 2247-2259.

McPhail, J.D. 2007. Freshwater Fishes of British Columbia. University of Alberta Press. Edmonton, Alberta.

Mitchell, P., Prepas, E.E. 1990. Atlas of Alberta Lakes. University of Alberta Press, Edmonton

Ney, J.J., Moore, C.M., Tisa, M.S., Yurk, J.J. and Neves, R.J. 1990. Factors affecting the sport fishery in a multiple-use Virginia reservoir. Lake and Reservoir Management 6:21-32.

Nurnberg, G.K., Peters, R.H. 1984. Biological availability of soluble reactive phosphorus in anoxic and oxic fresh-waters. Canadian Journal of Fisheries and Aquatic Sciences 4: 757-765.

Nurnberg, G.K. 2007. Lake responses to long-term hypolimnetic withdrawal treatments. Lake and Reservoir Management 23: 388-409.

Peterson, B.J., Deegan, L., J. Helfrich. J. Hobbie, M. Hullar, B. Moller, T. E. Ford, A. Hershey, A. Hiltner, G. Kipphut, M. A. Lock, D. M. Fiebig, V. McKinley, M. C. Miller, J. R. Vestal, R. Ventullo and G. Volk . 1993. Biological responses of a tundra river to fertilization. Ecology 74: 653-672.

Poff, N.L., Ward, J.V. 1989. Implications of streaflow variability and predictability for lotic community structure –a regional analysis of stream flow patterns. Canadian Journal of Fisheries and Aquatic Sciences 55: 201-217.

Power, M.E., Dietrich, W.E., Finlay, J.C. 1996. Dams and downstream aquatic biodiversity: potential food web consequences of hydrologic and geomorphic change. Environmental Management 20: 887-895.

Prchalova, M., Kubecka, J., Riha, M., Litvin, R., Cech, M., Frouzov, J., Hladik, M., Hohausova, E., Peterka, J., Vaskek, M. 2008. Overestimation of percid fishes (Percidae) in gillnet sampling. Fisheries Research 91:79–87.

Prchalova, M., Kubecka, J., Riha, M., Mrkvicka, T., Vasek, M., Juza, T., Kratochvil, M., Peterka, J., Drastik, V., Krizek, J. 2009. Size selectivity of standardized multimesh gillnets in sampling coarse European species. Fisheries Research 96: 51–57

Prchalova, M., Mrkvicka, T., Kubecka, J., Peterka, J., Cech, M., Muska, M., Kratochvil, M., Vasek, M. 2010. Fish activity as determined by gillnet catch: A comparison of two reservoirs of different turbidity. Fisheries Research 102: 291–296.

Randall, R.G. 1992. Using allometry with fish size to estimate production and biomass (P/B) ratios of salmonid populations Ecology of Freshwater Fish 11: 196-202.

Randall, R.G., Kelso, J.R.M, Minns, C.K. 1995. Fish production in freshwaters: are rivers more productive than lakes. Canadian Journal of Fisheries and Aquatic Sciences 52: 631-643.

Rasmussen, J.B. 1988. Littoral zoobenthic biomass in lakes, and its relationship to physical, chemical, and trophic factors. Canadian Journal of Fisheries and Aquatic Sciences 45:1436-1447.

Rawson, D.S. 1951. Studies of the Fishes of Great Slave Lake. J. Fish. Res. Board Can. 8: 207-240.

Redfield, A.C. 1958. The biological control of chemical factors in the environment. American Scientist 46: 205-221.

Ryan, P.M. 1984. Age and growth and food of threespine stickleback in three lakes of central Newfoundland, Canada. Canadian Technical Report Fisheries and Aquatic Sciences 1325 iv + 16 p.

Ryan, P.M. 1986. Lake use by wild anadromous atlantic salmon as an index of subsequent adult abundance. Canadian Journal of Fisheries and Aquatic Sciences 43: 2-11.

Shortreed, K.S., Morton, K.F., Malange, K., Hume, J.M.B. 2001. Factors limiting juvenile sockeye production and enhancement potential for selected B.C. nursery lakes. Canadian Science Advisory Secretariat Research Document 2001/098. 54 pp.

Smokorowski, K. E., Bergeron, N., Boisclair, D., Clarke, K., Cooke, S., Cunjak, R., Dawson, J. 2011. NSERC 's HydroNet: A National Research Network to Promote Sustainable Hydropower and Healthy Aquatic Ecosystems. Fisheries, 36(10), 480-488.

Stanley, E.H., Doyle, M.W. 2002. A geomorphic perspective on nutrient retention following dam removal. BioScience 52: 693-701.

Stockner, J.G., Shortreed, K. 1978. Enhancement of autotrophic production by nutrient addition in a coastal rainforest stream on Vancouver Island. Journal of the Fisheries Research Board of Canada. 35: 28-34.

Stockner, J., Langston, A., Sebastian, D., Wilson, G. 2005. The Limnology of Williston Reservoir: British Columbia's largest lacustrine Ecosystem. Water Quality Res. Journal of Canada. 40: 28-50.

Tilman, D., Downing, J.A. 1994. Biodiversity and stability of grasslands. Nature 367:363-65.

Van Zyll de Jong, I.C., Cowx, I.G., Scruton, D.A. 2005. Association between biogeographic factors and boreal lake fish assemblages. Fisheries Management and Ecology 12: 189-199.

Verdonschot, P.F.M. 2000. Integrated ecological assessment methods as a basis for sustainable catchment. Hydrobiologia 422: 389-412.

SECTION C - THERMAL REGIMES: Modelling water temperatures in regulated rivers

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DESCRIBING THE THERMAL REGIME OF RIVERS

Water temperature plays an important role in freshwater ecology and changes in temperature can significantly impact fish distribution, growth, mortality, production, habitat use and community dynamics (Caissie 2006). Most aquatic organisms are adapted to a certain range of temperature and organisms undergo thermal stress above a certain physiological tolerance threshold. For example, salmonids are particularly sensitive to warm water temperatures and considerable work has been done to determine upper thermal tolerance limit for salmonid species (Wehrly et al. 2007, Elliott and Elliott 2010, Breau 2013). In addition, seasonal and daily variations in water temperature are also important to consider given organisms have specific temperature requirements throughout their life history (Ward and Stanford, 1982, McCullough, 1999). Overall, the thermal regime of a river, which can be defined as the water temperature conditions in a river and how these vary through time, is of critical importance to maintain the ecological integrity of aquatic ecosystems (Olden and Naiman 2010). Accordingly, the characterization of the thermal regime of rivers is important in order to describe desirable conditions over time, compare different systems and assess anthropogenic impacts. Different conceptual frameworks have been proposed to describe the thermal regime of rivers.

Inspired from the natural flow regime paradigm (Poff et al. 1997), Olden and Naiman (2010) proposed to describe the thermal regime of a river by characterizing its five constituting components: magnitude, frequency, duration, timing and rate of change. Figure C1 provides examples of the five components of the thermal regime for a typical annual time series. The thermal regime can be summarized by using descriptive statistics which measure the central tendency or dispersion of each of its five components. Different series of descriptive metrics have been proposed to quantify the thermal regime of rivers (Chu et al. 2010, Guillemette et al. 2011, Arismendi et al. 2013, Rivers-Moore et al. 2013). In the context of impact assessments, this approach has the advantage of being able to target the specific components of the thermal regime that could be modified by the presence of a dam on the river and develop a management response accordingly. Although this approach allows for a comprehensive characterization of the thermal regime, it requires numerous metrics to capture the multivariate properties of water temperature variability and faces the inherent difficulty of selecting the most relevant and informative ones.

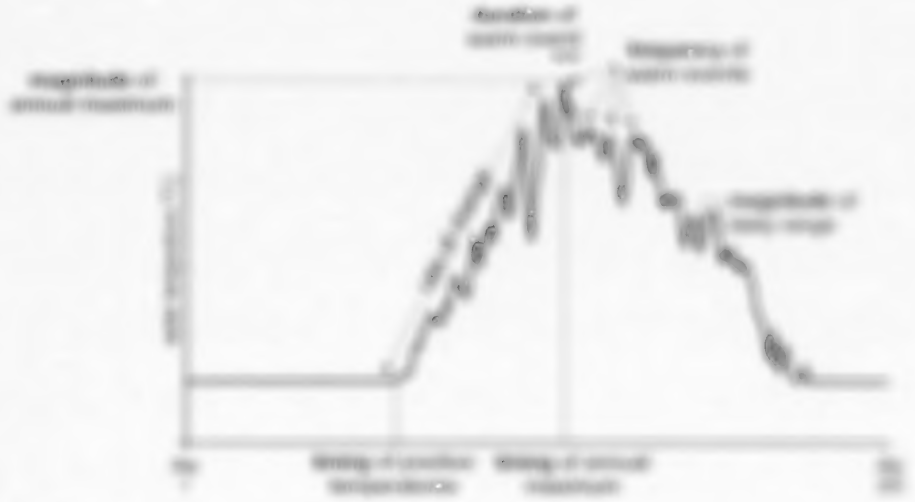


Figure 2.1. Examples of the five components (magnitude, frequency, duration, timing and rate of change) of a typical annual time series of river water temperature.

In the natural thermal regime approach described above, a large number of metrics are typically used to describing the annual cycle in water temperature, although metrics have also been proposed to describe variations at other time scales (e.g. daily, interannual). At northern latitudes, river water temperature generally follows a truncated sinusoidal function during the open water period. As such, another approach to the characterization of the thermal regime of rivers is to describe the annual water temperature cycle with a cosine function (Mathieu et al., submitted). The annual cycle can then be described by three parameters describing its magnitude (α_0), amplitude (A) and timing (ϕ):

$$T_w(t) = \alpha_0 + A\cos\left(\frac{2\pi}{365}(t - \phi)\right) \quad (1)$$

where t is the day of the year, α_0 is the annual average of the time series (T_w), A is the amplitude of the signal (T_w) and ϕ is the phase of the signal which corresponds to day of the year of the annual maximum. This approach offers an alternative to the characterization of the thermal regime with numerous and often redundant metrics by providing a parsimonious description of the annual water temperature cycle river using only three parameters. This approach can also be used to assess and describe the alteration of the annual water temperature cycle by dams or other anthropogenic activities by comparing the annual cycle, as described by the magnitude (α_0), amplitude (A) and timing (ϕ) parameters, between regulated and reference sites. For example, deviation of the magnitude parameter from reference conditions would result in a vertical translation of the annual signal of water temperature while deviation in the timing parameter would result in a horizontal translation of the annual signal (Figure C2). A difference in the amplitude parameters would lead to different warming/cooling rates in the spring/autumn as well as a difference in peak temperatures in the summer (Figure C2).

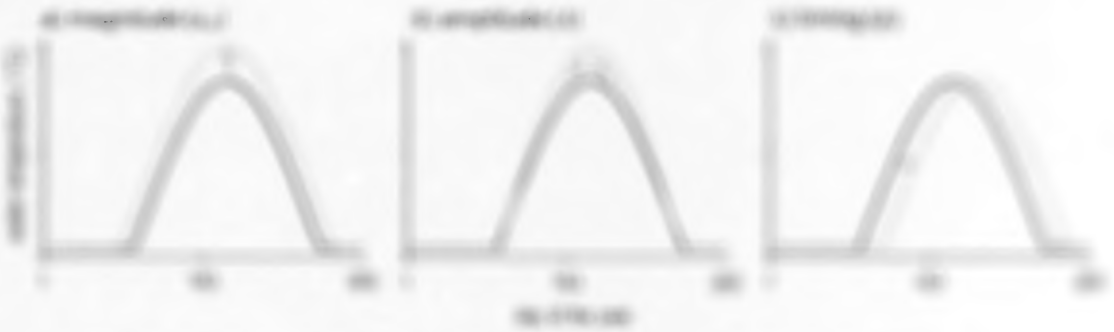
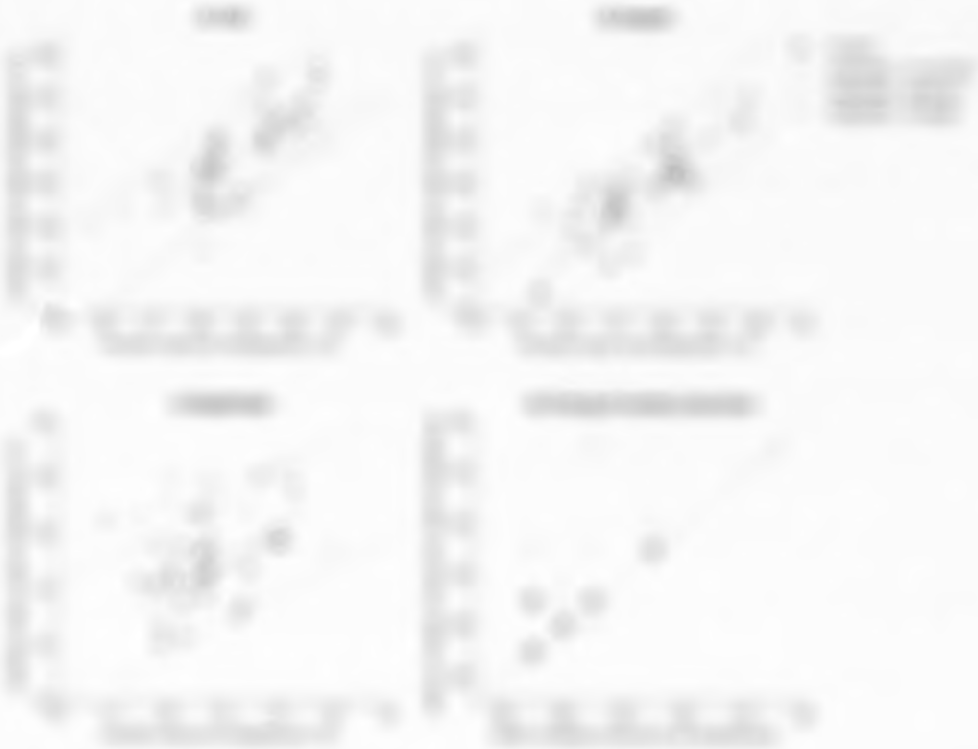


Figure 4.5. Schematic representation of changes in a) magnitude, b) amplitude and c) timing parameters describing the annual water temperature cycle in rivers

OBSERVED IMPACTS OF DAMS ON THE THERMAL REGIME OF RIVERS

The two approaches for the characterisation of the thermal regime have been used to compare the thermal regime of regulated and natural rivers. Using a series of metrics describing the magnitude, frequency, duration, timing and rate of change in summer water temperature conditions, a regional comparison of the summer thermal regime of natural versus regulated rivers has been performed for medium-sized rivers (drainage area of river upstream location between 4 700 and 6000 km²) in Eastern Canada (Ontario, Québec and New Brunswick). Regulated rivers represented three different operational types (run-of-river, storage and peaking). This regional analysis provides a statistical approach to quantitatively conceptualise the impact of dams on the thermal regime of rivers according to their operational type.

Water temperature was monitored by deploying two to ten thermal loggers along a river reach in regulated rivers. Thermal loggers were deployed as close to the dam as possible (within 0.1 to 1.5 km of the dam, with the exception of two sites where loggers were at 6.5 km and 9.7 km from the dam) and covered a distance ranging between 4 and 74 km (mean = 19 km) downstream of the dam. In natural rivers, thermal loggers covered a longitudinal distance ranging between 1 and 46 km (mean = 19 km). Metrics describing the magnitude, frequency, duration, timing and rate of change in summer water temperature were computed for each logger and the median value of the monitored reach was used to make comparisons among rivers. To allow comparison of thermal regimes at a regional scale and for different years, air temperature conditions were taken into consideration when describing the magnitude and timing of water temperature regimes. Figure 4.6 shows the relationship between air and water temperature metrics for 29 rivers (17 natural and 12 regulated rivers) for the summ, rivers at 2011 to 2013. Each point on the figure represents conditions in one river in a given year and the dotted line is used as a 1:1 and 1:1 represents the 95% confidence interval of the linear model between air and water temperature metrics for natural rivers. As illustrated in Figure 4.6, the presence of a reservoir (i.e. storage and peaking dams) led to considerable differences in the thermal regime of rivers, particularly regarding the magnitude of the thermal regime in September and the timing of the weekly maximum water temperature in the summer. For example, water temperatures were generally warmer in September in regulated rivers with storage or peaking operations (Figure 4.6c). The warming effect of storage and peaking dams was observed throughout the entire monitored reach of the regulated rivers (1 to 74 km). In fact, these distances should be considered as the minimum distance influenced and it is likely that the warming effect extended over greater distances downstream, but data were not available further away from the dam. The magnitude of the thermal regime of storage and peaking systems was influenced to a lesser extent in July and August, although storage



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2014). The deterministic model used was SNTEMP (Bartholow, 1989), a simulation tool that has often been used in the United States, to investigate the impacts of impoundments on river temperature. The statistical model is based on an interpolation method implemented in a multivariate space created using canonical correlation analysis (Guillemette et al. 2011). Both models were calibrated on two reaches of the Fourchue River, upstream and downstream of the reservoir. SNTEMP was used to simulate mean water temperatures time series using meteorological inputs and discharge. The statistical model was used to directly estimate thermal indices (descriptive statistics of the thermal regime). The two models were compared based on their efficiency to estimate thermal indices such as mean and maximum monthly water temperature. Temperature was monitored at 18 locations in the Fourchue River during the summers of 2011 and 12 locations in 2012 to describe the thermal regime and calibrate the models. Both models were able to simulate mean monthly thermal indices adequately (e.g. Figure C6). However, the statistical model achieved better results than SNTEMP in estimating some of the thermal indices (Figure C7), especially the mean and maximum daily ranges with Root Mean Square Errors (RMSE) of 4.1 °C and 4.9 °C respectively for SNTEMP as compared to 0.6 °C and 1.4 °C for the statistical model.

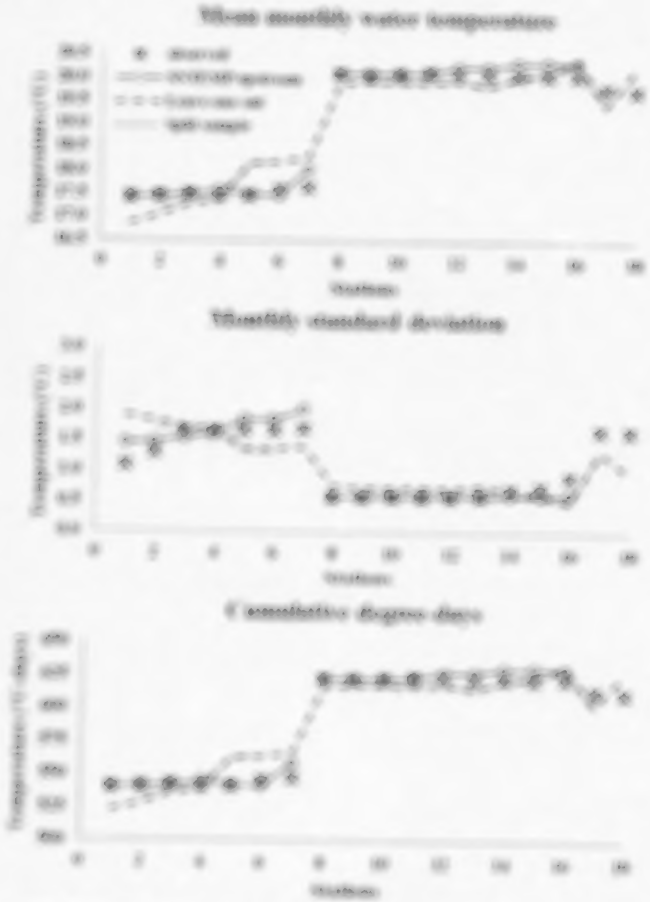

Figure C6. Observed and simulated mean monthly water temperature, standard deviation and cumulative degree-days for August 2011, using SNTEMP and the statistical model in Leave-one-out and split-sample modes. Stations 1 to 18 are from upstream to downstream (from Beaupré, 2014).

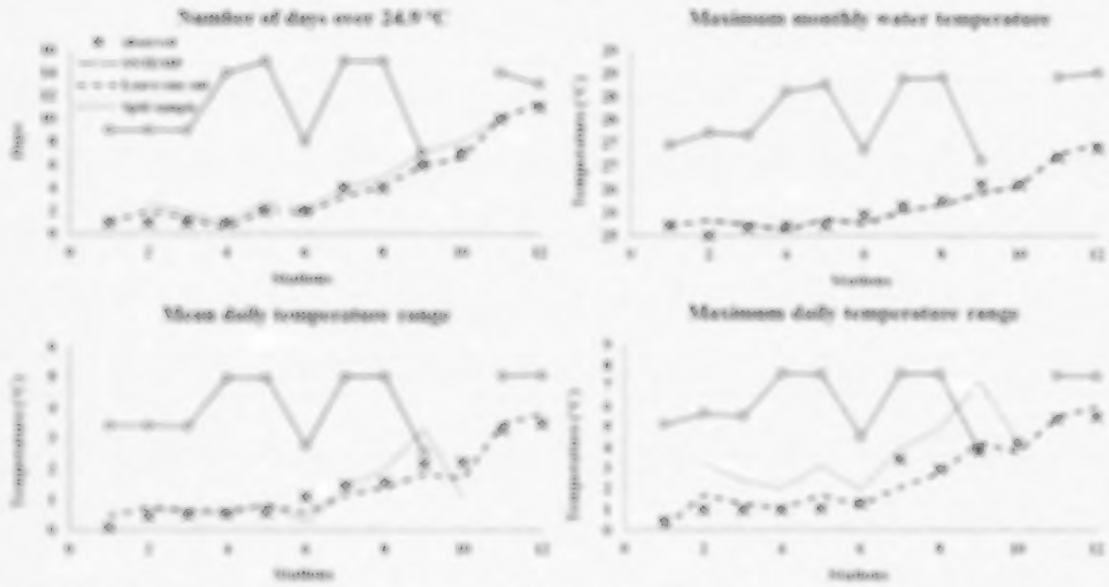

Figure C7. Observed and simulated monthly maximum water temperatures, mean and maximum temperature ranges and the number of days over 24.9 °C for August 2012, using SNTEMP and the statistical model in Leave-one-out and split-sample modes. Stations 1 to 12 are from upstream to downstream (from Beaupré, 2014).

Subsequently, additional models were added to the comparative study, with the condition that they all use at least both air temperature and flow as input data. The inclusion of flow as a required input ensured that all models could be used in the context of the presence of a dam, to study different flow management scenarios. In the deterministic category, the CEQUEAU hydrological and water temperature model was used (Morin and Couillard, 1990). Three statistical models were also implemented: an Artificial Neural Network (ANN), an Auto-regressive Moving Average with eXogenous variables model (ARMAX) and a Nonlinear Autoregressive with eXogenous variables model (NARX). The CEQUEAU model is a semi-distributed model that has been recently re-programmed by Rio-Tinto Alcan. The hydrological component of CEQUEAU takes into account the physical characteristics of the watershed by decomposing it in Elementary Representative Areas (ERA) of equal surface for which the altitude, percentage of forest cover and the percentage covered by lakes and wetlands are defined. On each ERA, vertical routing of water is simulated daily, starting with meteorological inputs (total daily rain and/or snow), and through reservoirs representing the upper and lower soil horizons. Detailed description of both vertical and upstream-downstream routing of water by CEQUEAU was given by Morin and Couillard (1990). The water temperature module was described by St-Hilaire et al. (2000). Water temperature is defined by calculating the ratio of enthalpy (H) over the product of water volume (V, as simulated by the hydrological module) and water specific heat capacity (Θ). Changes in the enthalpy are calculated for each ERA, and each day using a heat budget approach. The semi-empirical relations used in the heat budget were all described by Morin and Couillard (1990).

A comparison of these models was completed for the ice-free periods of 2011 to 2014. Calibration of the deterministic model was done by hand, with a split sample approach. Table C1 summarizes the performance statistics of the four models. Except for the ANN, all models performed relatively well, with RMSE of the order of or less than 1°C. The CEQUEAU

model showed a relatively strong negative bias for the calibration phase, caused by the fact that it was not as efficient as the statistical models to reproduce higher water temperatures. Given its autoregressive nature, the ARMAX model outperformed the other statistical approaches, while the ANN performed more poorly, most likely because more data would be required for the learning phase of this non-parametric approach.

Table C1. Root Mean Square Error (RMSE) and bias calculated between observed and simulated water temperatures at a station located 7 km downstream of the Morin Dam on the Fourchue River (From Kwak et al., 2016).

Model	Calibration phase		Validation phase	
	RMSE (°C)	BIAS (°C)	RMSE (°C)	BIAS (°C)
CEQUEAU	1.03	-1.17	1.05	-0.74
ANN	1.86	0.54	2.53	7.14
ARMAX	0.62	0.02	0.96	1.28
NARX	0.71	0.26	1.05	0.94

All models used in this comparative study could potentially be used to study the impact of the management of water releases associated with dam operations on a river if difficulties in the calibration process are overcome. Large data requirement is often stated to be an impediment to the implementation of deterministic models. Both SNTEMP and CEQUEAU require information about basin physiography and land use. In addition, SNTEMP requires information about canopy closure above the stream. However, both deterministic models allow for simulation of temperatures at different locations of the river reach, and for different dam water release scenarios. The only tested statistical model that is not local (i.e. limited to one station) is the geostatistical model of Beaupré (2014). While the other statistical models are 0-D (i.e. only provide simulated water temperatures at one point on the river). The geostatistical approach used by Beaupré (2014) can provide estimates anywhere along the river, provided that there is a sufficient number of monitoring stations to perform the initial interpolation in multivariate space.

Hence, it can be seen that there are some alternatives to deterministic approaches to simulate the impact of dams on the thermal regime. Model selection should be based on the following criteria:

1) data availability;
2) model capacity to account for flow variability;
3) the requirement to simulate water temperature at different locations; and
4) the relative ease of calibration and validation.

MANAGING THE THERMAL REGIME OF REGULATED RIVERS

The natural dynamics of water temperature are important to maintain the integrity of rivers. Yet, seasonal and daily patterns of variations in water temperature previous to dam construction are often unknown given the common absence of pre-dam data. In the absence of information on the thermal regime of a river previous to dam construction, different approaches are available to determine the desirable natural thermal regime template. Empirical or statistical models based on unregulated rivers can be used to estimate the natural thermal regime and help set management goals in regards to water temperature. For example, the magnitude and timing of the natural thermal regime of a river can be broadly estimated from air temperature measurements (Figure C3). Care should be taken when using regional models given the

relation between air and water temperature conditions varies according to the size and position of the river in the watershed. Our ability to model regional patterns of sub-daily variations in water temperature remains limited, although a good understanding of the range of values observed in natural rivers can help define a range of acceptable values in regulated rivers (Figure C4).

Various statistical and deterministic models are also available to simulate water temperature in regulated rivers. As shown in Section 3, many of these models can be adapted to account for flow variability and as such can be used to study different flow management scenarios. In addition, statistical and deterministic models can be used to study the spatial extent of a dam's impact on the thermal regime of rivers.

In addition to considering a natural regime template, water temperature in regulated rivers can also be managed according to life history requirements of species of interest. Through the characterization of the annual water temperature cycle, the full life cycle needs of a species can be assessed. For example, the thermal regime of the Fourchue River (storage dam) would not provide optimal growth conditions for brook trout given daily mean water temperature would exceed 19°C for more than 60 days according to the estimated annual water temperature cycle (Figure C8). In addition, a year-around perspective on river water temperature also highlights the early accumulation of thermal units in the Fourchue River as a result of the warming effect of the dam during the winter (Figure C6). For example, mean daily water temperature ranged between 1.2 and 3.3°C during the winter and as such, the thermal assessment would suggest further investigation into a potentially accelerated development of aquatic organisms in the warmer regulated river.

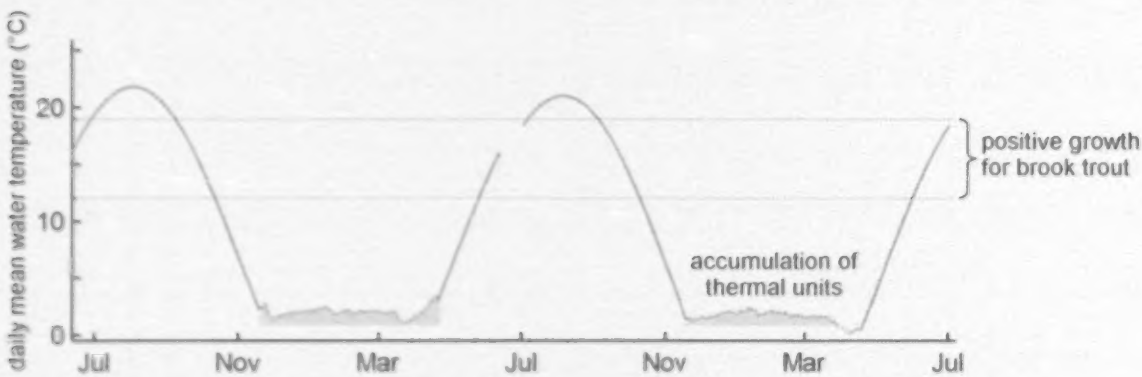

Figure C8. Annual cycle in daily mean water temperature for the Fourchue River (storage dam) and potential influence on life history and bioenergetics.

LITERATURE CITED

Arismendi I, Johnson SL, Dunham JB, Haggerty R. 2013. Descriptors of natural thermal regimes in streams and their responsiveness to change in the Pacific Northwest of North America. Freshwater Biology: 58(5):880-894.

Bartholow, JM. 1995. The stream network temperature model (SNTEMP): A decade of results. Workshop on Computer Application in Water Management. Fort Collins, CO: Water Resources Research Institute, CSU. p. 57-60.

Beaupré, L. 2014. Comparaison de modèles thermiques statistique et déterministe pour l'estimation d'indices thermiques sur les portions aménagées et naturelles de la rivière Fourchue (Québec, Canada). M. Sc. Thesis, INRS-ETE. 61 p.,

Benyahya L, Caissie D, St-Hilaire A, Ouarda TBMJ. 2007. A review of statistical water temperature models. Canadian Water Resources Journal, 32(3):179-192.

Breau, C. 2013. Knowledge of fish physiology used to set water temperature thresholds for in-season closures of Atlantic salmon (*Salmo salar*) recreational fisheries. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/163.

Caissie D. 2006. The thermal regime of rivers: a review. Freshwater Biology 51: 1389–1406.

Caissie, D., Breau, C., Hayward, J., and Cameron, P. 2013. Water temperature characteristics within the Miramichi and Restigouche rivers. DFO Can. Sci. Advis. Sec. Res. Doc. 2012/165.

Chu C, Jones E, Allin L. 2010. Linking the thermal regimes of streams in the Great Lakes Basin, Ontario, to landscape and climate variables. River Research and Applications 26: 221–241.

Daigle A, St-Hilaire A, Beveridge D, Caissie D, Benyahya L. 2011. Multivariate analysis of the low-flow regimes in eastern Canadian rivers. Hydrological Sciences Journal 56: 51–67.

Elliott JM, Elliott J a. 2010. Temperature requirements of Atlantic salmon Salmo salar, brown trout Salmo trutta and Arctic charr Salvelinus alpinus: predicting the effects of climate change. Journal of fish biology 77: 1793–817.

Guillemette N, St-Hilaire A, Ouarda TBMJ, Bergeron N. 2011. Statistical tools for thermal regime characterization at segment river scale: Case study of the Ste-Marguerite River. River Research and Applications 27: 1058–1071.

Kwak, J. A. St-Hilaire, F. Chebana. 2016. A comparative study for water temperature modelling in small basin, the Fourchue River, Quebec, Canada. Hydrological Sciences Journal. (in press)

Maheu A, Poff NL, St-Hilaire A. 2015. A classification of stream water temperature regimes in the conterminous United States. River Research and Applications. DOI: 10.1002/rra.2906

McCullough DA. 1999. A review and synthesis of effects of alterations to the water temperature regime on freshwater life stages of salmonids. : 279.

Morin G, Couillard D. 1990. Predicting river temperatures with a hydrological model. In Chereminisoff NP. (ed) Encyclopedia pf fluid mechanics, surface and groundwater flow phenomena (pp.171-209). Gulf Publishing Company: Huston, USA.

Olden JD, Naiman RJ. 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshwater Biology 55: 86–107.

Poff NL, Allan JD, Bain MMB, Karr JJR, Prestegaard KL, Richter BDB, Sparks RE, Stromberg JJC, KL P, RE S. 1997. The Natural Flow Regime. Bioscience: 769–784.

Rivers-Moore N a, Dallas HF, Morris C. 2013. Towards setting environmental water temperature guidelines: a South African example. Journal of environmental management 128: 380–92.

Simpkins DG, Hubert WA, Wesche TA. 2000. Effects of fall-to-winter changes in habitat and frazil ice on the movements and habitat use of juvenile rainbow trout in a Wyoming tailwater. Transactions of the American Fisheries Society 129: 101–118.

St-Hilaire A, Morin G, El-Jabi N, Caissie D. 2000. Water temperature modelling in a small forested stream: implication of forest canopy and soil temperature. Canadian Journal of Civil Engineering, 27:1095-1108.

Ward J V, Stanford JA. 1982. Thermal responses in the evolutionary ecology of aquatic insects. Annual Reviews in Entomology 27: 97–117.

Wehrly KE, Wang L, Mitro M. 2007. Field-Based Estimates of Thermal Tolerance Limits for Trout: Incorporating Exposure Time and Temperature Fluctuation. Transactions of the American Fisheries Society 136: 365–374.